Institute for European Environmental Policy

ESTIMATION OF THE FINANCING NEEDS TO IMPLEMENT TARGET 2 OF THE EU BIODIVERSITY STRATEGY

REVISED DRAFT FINAL REPORT

ENV.B.2/ETU/2011/0053r

24th July 2013

Institute for European Environmental Policy (IEEP)

London office:

11 Belgrave Road
London SW1V 1RB
United Kingdom

Brussels Office:

Quai au Foin, 55
Hooikaai 55
1000 Brussels
Belgium

Contact person:

Graham Tucker
11 Belgrave Road
London SW1V 1RB, UK
United Kingdom
Email: gtucker@ieep.eu

The Institute for European Environmental Policy (IEEP) is an independent, not for profit institute dedicated to advancing an environmentally sustainable Europe through policy analysis, development and dissemination. Based in London and Brussels, the Institute's main focus of research is on the development, implementation and evaluation of EU policies of environmental significance, including agriculture, biodiversity, climate and energy, fisheries, industrial policy, regional development, transport, waste and water.
See www.ieep.eu for further details

Authorship

The recommended citation for this report is: Tucker, Graham; Underwood, Evelyn; Farmer, Andrew; Scalera, Riccardo; Dickie, Ian; McConville, Andrew; van Vliet, Wilbert. (2013) Estimation of the financing needs to implement Target 2 of the EU Biodiversity Strategy. Report to the European Commission. Institute for European Environmental Policy, London.

Additional contributions were received from:

Ben Allen, Jane Desbarats, Marianne Kettunen, Christina Ieronymidou, Leonardo Mazza, Stephanie Newman, Caitlin McCormack, James Weaver and Guy Whiteley.

Disclaimer

Acknowledgements

We particularly thank the following reviewers for comments on the draft report and information on the costs of maintaining and restoring ecosystems: Kris Decleer, Sabine Tischew, Katalin Török We are also grateful to Bent Jepsen of Astrale and his colleagues for assistance with examination of LIFE Nature project information on restoration costs. Philine zu Ermgassen of Cambridge University provided helpful insights on the challenges of identifying appropriate restoration baselines.

We are grateful to Allan Buckwell and Jana Poláková at IEEP for advice on the study and comments on the draft report, and Sarah Green for assistance with proof reading and copy-editing.

Lastly, we thank the Project Steering Committee at the European Commission and in particular the project Desk Officers Szilvia Bosze and Patrick Murphy for their guidance during the study, and Marco Fritz for help with setting up the assessment of LIFE project data.

Contents

LIST OF ACRONYMS

AES	Agri-environment scheme
BAP	Biodiversity Action Plan
CAP	Common Agricultural Policy
CBD	UN Convention on Biological Diversity
CFP	Common Fisheries Policy
CIP	Competitiveness and Innovation framework Programme
EAFRD	European Agricultural Fund for Rural Development
EEA	European Environment Agency
HNV	High Nature Value
IAS	Invasive Alien Species
LFA	Less Favoured Area Payments
MFF	Multi-annual Financial Framework (with respect to the EU)
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
PES	Payments for Ecosystem Services
PoM	Programme of Measures (with respect to the Water Framework Directive)
PPPs	Public Private Partnership schemes
RBMP	River Basin Management Plans
RDP	Rural Development Programmes
RFF	Risk-Sharing Finance Facility
SEA	Strategic Environmental Assessment
SME	Small or Medium Enterprise
UWWTD	Urban Waste Water Treatment Directive
WFD	EU Water Framework Directive

EXECUTIVE SUMMARY

Background and aims of this study

The EU has a target to "halt biodiversity and ecosystem service loss by 2020, to restore ecosystems in so far as is feasible, and to step up the EU contribution to averting global biodiversity loss". To help achieve this target an EU 2020 Biodiversity Strategy has been developed by the European Commission and endorsed by the Environment Council that includes six interrelated subtargets and further supporting actions. This study focuses on Target 2 of the Biodiversity Strategy, which is that "***By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems***[1]". Target 2 is similar to, and aims to support, Target 15 of the 2020 Strategic Plan of the Convention of Biological Diversity. Target 2 is also supported by three key actions, which aim to improve knowledge of ecosystems and their status, develop a green infrastructure for the EU and ensure no net loss of biodiversity.

In the context of Target 2 and this report, restoration is defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed"[2]. Green infrastructure is defined by the European Commission in its 2013 Green Infrastructure Strategy as "*a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, GI is present in rural and urban settings*."

One of the main reasons for the adoption of the target is the increasing recognition of the multiple services that ecosystems provide to humankind, including economic, social and cultural benefits, as well as their intrinsic nature conservation values. Therefore, although the target is ambitious and will require new actions with associated costs, it is foreseen that the costs of achieving Target 2 will be offset by many benefits, especially actions that maintain ecosystem values and thereby avoid more difficult and costly restoration actions in future. Nevertheless, it will be necessary to secure increased resources (potentially from both private and public sources) for environmental management and restoration beyond those that already contribute to ecosystem maintenance and restoration. It is therefore necessary to quantify these additional costs and to identify potential means of financing them to inform debates on potential options for achieving the target effectively and efficiently.

This scoping study therefore aimed to provide an estimation of financing needs for implementing Target 2 of the EU 2020 Biodiversity Strategy, and in particular the costs of restoring at least 15% of degraded ecosystems. It also provides a general analysis of the main options for financing the achievement of the target, focusing

[1] ie 15% of those areas of each ecosystem type that are degraded

[2] Society for Ecological Restoration (2004)

especially on the extent to which private sources could complement financing from the main EU funding instruments. A particular focus is given to innovative sources of funding, and funding from the private sector.

To achieve these aims, this study firstly reviewed the results of previous studies that have attempted to calculate the costs and benefits of ecosystem maintenance and restoration in relation to various environmental objectives, such as the management of Natura 2000 sites, and restoration of water and soil related ecosystem services. This revealed, not unexpectedly, that previous assessments do not provide a complete or up to date picture of the likely costs and benefits of achieving Target 2. Information on the benefits of ecosystem restoration in the EU were found to be especially scarce or of little relevance to this study. In contrast there is a large amount of data on the costs of practical ecosystem maintenance and restoration costs, most notably from agri-environment schemes, as well as some nature conservation projects (such as those carried out under the EU LIFE Programme). Therefore most of this study entailed a new analysis of existing data on ecosystem degradation and the costs of maintaining and restoring ecosystems.

An important objective of this study was to estimate the costs of achieving Target 2 that are in addition to existing and expected expenditure on measures that are already contributing to the maintenance and restoration of ecosystems. Therefore all the costs calculated in this study are for annual costs in 2020 in relation to a **Reference Scenario** that takes into account forthcoming ecosystem conservation and restoration measures expected under existing legislation, including the Habitats and Birds Directives, and the Water Framework Directive (WFD), as well as anticipated funding of land management actions such as through the Common Agricultural Policy (CAP).

This study considered three main types of activity that are required to achieve Target 2:

- supporting actions listed in the Biodiversity Strategy;
- measures that treat the source of generic wide-scale pressures (such as water and air pollution) ; and
- practical ecosystem management, restoration and re-creation measures (such as hydrological management, grazing, removal of invasive species and vegetation re-establishment).

As these types of action differ considerably in their scale, duration and likely cost, they are assessed separately, as summarised below. This is followed by a summary of the benefits of restoration and potential financing instruments for restoration and green infrastructure.

The costs of supporting actions listed in the Biodiversity Strategy

The supporting actions listed in the Biodiversity Strategy under Target 2 are improving knowledge of ecosystems and their services (Action 5), promoting a green infrastructure (Action 6) and ensuring no-net loss of biodiversity (Action 7). Action 5 is currently underway through the Mapping and Assessing Ecosystem Services (MAES) initiative, which is principally a research project. Whilst it is an ambitious and large-scale initiative, its costs are in a different order of magnitude to the costs of more practical measures described below, and were therefore not considered in this study.

Action 6 includes two sub-actions that will be led by the Commission. Action 6a is to develop with Member States a strategic framework to set priorities for ecosystem restoration at sub-national, national and EU levels. The cost of this action is expected to be very limited as it primarily involves the development of tools by the Commission, with support from a contractor.

Action 6b was to develop an EU level Green Infrastructure Strategy, which the Commission completed in May 2013. The strategy envisages that the EU policy action will result in an '*enabling framework*' providing a combination of policy signals and technical or scientific actions, implemented within the context of existing legislation, policy instruments and funding mechanisms. The costs of implementing this strategy (involving a large number of specific decisions) are therefore likely to relate to overarching measures to support actions on the ground, as well as more practical measures. Support measures may include providing information on the benefits of green infrastructure, development of mapping tools, and guidance to relevant authorities. Such actions are likely to cost in the order of several million Euros, but are likely to overlap with existing initiatives and may therefore not be entirely additional costs. Such costs are therefore likely to be very small in comparison to practical restoration measures.

Practical actions to implement the strategy would be expected to include measures to maintain, restore and re-create ecosystems and their components. However, it is very difficult to quantify such as actions as the target is not defined in a quantitative way. Furthermore, many actions would be expected to overlap with existing initiatives and new ecosystem restoration measures that would be required to achieve Target 2, as described below. Therefore, to avoid the double-counting of costs, the costs of practical measures to implement the Green Infrastructure Strategy are not calculated separately, but are assumed to be captured in the practical actions as estimated below.

It is very difficult to estimate the costs of implementing a no net loss policy in the EU (Action 7) as its scope and means of implementation are uncertain. Nevertheless, it seems likely that some form of biodiversity offsetting will be required to compensate for unavoidable residual impacts, in which case the main costs will be borne by project proponents. Some public funding would also be required to support the institutional capacity required for regulating and advising on no net loss measures (such as offsets). The private and public costs of the no net loss policy initiative will depend greatly on whether offsetting is to be mandatory, its sectoral scope (eg whether it covers agricultural and forestry activities) and the policy and legislative framework that is promoted under the initiative. Despite this uncertainty it seems unlikely that the additional costs of Action 7 will be significant compared to wider Target 2 costs, especially as one of the principal aims of the policy is to avoid and reduce residual damage as much as possible, such that offsetting is normally carried out as a last resort.

Costs of wide-scale measures

It is evident from a number of monitoring programmes and research studies that some of the most widespread and significant impacts on ecosystems and their associated species result from water and air pollution. These pressures will therefore need to be tackled if Target 2 is to be achieved. However, such pressures cannot normally be easily dealt with at their point of impact, but generally require measures that reduce pollution emissions at their source. This study therefore examined the implementation of existing measures that are being taken to reduce pollutant emissions and attempted to establish if such measures would be sufficient to achieve Target 2. However, as such measures are wide ranging and aim to tackle a variety of environmental problems it is not possible to ascertain the proportion of costs that might be attributed to achieving Target 2. Furthermore, as sophisticated modelling would be required that is beyond the scope of this study, it was not possible to estimate the additional costs of addressing any shortfalls in measures with respect to the achievement of Target 2.

This study found that there are still many significant pressures on Europe’s aquatic ecosystems. However, the Urban Waste Water Treatment Directive, Nitrates Directive and more recently the WFD provide strong legal frameworks to achieve ecosystem restoration and maintenance of Europe’s waters if properly implemented. Moreover, requirements under the WFD to prevent the deterioration of waters and to achieve Good Ecological Status or Good Ecological Potential appear to be broadly equivalent to Target 2 requirements for maintenance and restoration.

Although Member State reporting indicates that progress with the implementation of the WFD is slower than expected, evidence indicates that more than 15% of water bodies in the EU that do not currently have a good status will be restored to good status by 2020. Therefore, it seems likely that Target 2 restoration requirements related to water quality (and some other key ecological attributes of the water body), will be met and there will be no additional Target 2 related costs over and above those entailed in complying with the WFD and older Directives.

Air pollution is a threat to natural and semi-natural ecosystems, as a result of acidification (from deposition of sulphur dioxide), nutrient enrichment (ie deposition of nitrogen compounds on sensitive ecosystems), and regionally high levels of toxic ozone. However, current legislation and policy is delivering significant air quality improvements with reductions in all of the above pollutants. Furthermore, monitoring and models indicate that there will be a greater than 15% reduction in the overall area of ecosystems affected by acidification in the EU by 2020, and therefore the achievement of the restoration component of Target 2 will be achieved in this respect with no additional costs. But it should be noted that some sensitive ecosystems may still be impacted significantly and therefore the maintenance component of Target 2 may not be fully met.

Despite some progress towards the target, it seems highly unlikely that current legislation would lead to the EU achieving the 15% target with respect to air-borne eutrophication. To address this, Target 2 requirements need to be considered within the analysis supporting the current review of air pollution policy being undertaken by the Commission, and future air policy targets aligned accordingly.

It should also be noted that restoration of eutrophied and acidified ecosystems is not simply achieved by reducing pollutant loads. Although continuing damage is stopped, past damage is not necessarily restored. Therefore to meet Target 2 by 2020, proactive habitat management (such as removal of nutrient enriched soils) will be required, the cost implications of which are likely to be substantial but difficult to predict precisely.

Costs of practical ecosystem specific measures

Included costs and estimation methods

The largest additional costs of achieving Target 2 relate to practical ecosystem management and restoration and re-creation measures. Therefore this study attempted to provide a detailed estimate of the potential additional annual costs in 2020[3] of maintaining and restoring each main type of terrestrial, coastal and freshwater ecosystem type recognised in this study.

Estimates of the costs of practical maintenance and restoration measures were not calculated for marine ecosystems. This is because the maintenance and restoration of such ecosystems is primarily dependent on regulations (such as on fishing practices, developments at sea and control of pollutants) rather than proactive practical actions. Although some practical actions can help to restore marine ecosystems, Target 2 can probably be mostly achieved through improved regulation, achieving over time reductions in pressures that are sufficient to enable passive restoration through natural processes.

[3] This is done to facilitate comparison of maintenance, restoration and re-creation costs, by dividing total cumulative restoration and re-creation costs to 2020 by 10 (ie assuming measures are taken between 2010 and 2020).

The calculation of the additional costs of maintaining and restoring each ecosystem type to achieve Target 2 was carried out by:

- Estimation of the area of each ecosystem type in the EU (largely drawing on CORINE Land Cover data).

- The identification of **key pressures** (ie the most important pressures that need to be addressed in order to maintain and restore the ecosystem) affecting each ecosystem type and the percentage of the ecosystem significantly impacted.

- Estimation of overall levels of ecosystem degradation, with respect to:
    - 2010 baseline levels (or as close to 2010 as data allow); and
    - expected levels in 2020, taking into account existing policies and expected measures as well as anticipated changes in drivers of land use change (ie **the reference scenario**).

- Identification of **key measures** (ie the most effective and efficient measures that are typically used to address one or more of the key pressures) and their costs.

- Calculation of total ecosystem maintenance and restoration costs, according to two methods, using:
    - **overall rates of degradation** and the costs of addressing pressures through ecosystem-specific **combined measures** (ie packages of measures) that aim to maintain and restore the ecosystem; and
    - specific estimates of the extent of each **key pressure** on the ecosystem and the costs of specific **key measures** to address them.

The two methods are used in the last stage of the cost calculation to provide comparative estimates that may help to judge their reliability. However, it is considered that the use of overall degradation levels, together with estimates of generic combined costs, may be too simplistic and therefore provide unreliable estimates of costs, especially where there are considerable variations in the costs of key measures and their required use. Therefore the estimates based on specific key pressures and key measures are likely to be more reliable. However, for some ecosystem types insufficient data were available to enable calculations using both methods.

The concept of degraded ecosystems and their baseline condition

One issue that has a major influence on the results of this study is the interpretation of the term '**degraded**'. For ecosystems that primarily comprise natural and semi-natural habitats of Community importance it was assumed that they are degraded if their habitats have an unfavourable conservation status as defined and monitored in accordance with the EU Habitats Directive and related guidance. The assessment of degradation of artificial ecosystems is more problematical because the concept of favourable conservation status cannot be easily applied to them and, critically, they are not therefore monitored in this respect. As a result, this study did not attempt to define or quantify overall degradation levels for artificial ecosystems. Instead, maintenance and restoration costs were only based on the proportion of these ecosystems that are affected by specific key pressures (such as soil erosion). Furthermore, it was assumed for this study that appropriate restoration goals within artificial ecosystems are to maintain and restore biodiversity and key ecosystem services to acceptable levels for the ecosystem in question, rather than to fully restore them to original semi-natural or natural ecosystem types, which clearly is impractical.

Another issue that needed to be considered over the interpretation of Target 2 relates to the time period over which degradation should be measured, that is to say its **baseline**. The Target does not refer to any baseline date or cut-off date against which degradation should be measured. This might imply that all former areas of natural and semi-natural ecosystems that have been converted to artificial ecosystems should be regarded as degraded and therefore 15% should be subject to re-creation, but this is also clearly impractical. Therefore, for the purposes of this study it was assumed that the 15% restoration requirement under Target 2 refers to ecosystems that have been destroyed since 2000, as well as ecosystems that were considered to be degraded in 2010 according to the EEA's Biodiversity Baseline report.

The study was not able to assess some costs, most notably opportunity costs (eg relating to regulations that prevent land use intensification that could sustainably increase income for the landowner) unless they are compensated for through public funding mechanisms (such as agri-environment payments). Nor was it able to estimate the costs of supporting actions (such as general ecological research and monitoring, development of strategies, policies and legislation, advice and training, site wardening, regulatory enforcement and awareness raising on biodiversity issues), species-specific measures (eg relating to particular habitat needs or translocations) , specific measures to reduce habitat fragmentation beyond general habitat restoration, or land purchase requirements (although these probably need not be great to achieve Target 2). However, most of these supporting actions are already underway to some extent, and it is unlikely that additional costs to achieve Target 2 would be substantial in comparison to the costs of more practical ecosystem maintenance and restoration measures as estimated below.

Estimates of the costs of practical ecosystem maintenance, restoration and re-creation measures

It is important to note that the estimated costs of achieving Target 2 are **in addition to those associated with expected measures up to 2020** (ie the reference scenario). Cost estimates are the expenditure required in 2020, which comprise the costs of annual maintenance measures and the one-off costs of restoration and re-creation divided evenly over the 2010 to 2020 period. Costs are based on the current costs of measures and are not adjusted for anticipated inflation or other possible changes in the costs of measures in 2020. Minimum and maximum cost estimates are provided, which reflect the estimated minimum and maximum degradation levels (with respect to overall degradation levels and degradation resulting from specific key pressures).

The annual additional cost of **maintaining** the ecological condition of all ecosystems is estimated to be between **€618 and 1,660 million**. The costs of maintaining each ecosystem vary considerably, largely due to their differing area. But costs also vary in relation to the need for management measures and the extent to which such requirements are already provided for through existing and expected measures. Hence the highest maintenance costs are for arable and forest ecosystems, requiring approximately €220 to 556 million and €196 to 488 million respectively. The national costs of maintenance measures also varies considerably, principally in relation to the area of each Member State and the proportion of ecosystems within each that have costly maintenance requirements.

In the calculation of restoration costs (including the re-creation of ecosystems areas lost since 2000) it is necessary to allow for the fact that the 15% restoration target is a target for the EU as whole and can be potentially disaggregated so that, for example, a higher proportion of some ecosystems are restored than others. Therefore, to examine the biodiversity and ecosystem service impacts and the cost implications of disaggregating the overall target this study estimated the total restoration and re-creation costs for the EU as a whole and for each Member State according to five scenarios, as described below.

Scenario A was the default scenario with 15% restoration and re-creation for each ecosystem, and accordingly a detailed breakdown of the costs of achieving Target 2 for each ecosystem under this scenario is provided in the main report. Under this scenario, the total average annual **restoration** costs for 2020 range from **€7.79 to 10.9 billion**. Despite the even percentage restoration allocation across all ecosystems the majority of the costs would be associated with measures for arable ecosystems, as a result of their large area, high levels of degradation and the relatively high cost of restoration measures for them. **Re-creation** costs under this scenario would range from **€66 to 117 million**. The majority of the costs are for semi-natural grasslands and to a lesser extent sclerophyllous vegetation. When interpreting these results it should be borne in mind that there are no re-creation targets for arable, permanent crops or improved grasslands (as these artificial ecosystems do not merit re-creation), forests (because there has been a net increase in their area in recent decades) and mires, lakes and rivers (because they cannot normally be feasibly re-created with reasonable costs).

Scenario B attempts to reduce the cost of achieving Target 2 by focusing restoration measures on ecosystems that have the lowest unit cost for restoration (ie forests, heathland and tundra, mires, lakes and rivers (other than components covered by the WFD) and saltmarshes). This would result in a substantial reduction in costs, with the total for **restoration** coming down to **€506 million to €1.75 billion** and re-creation costs falling to €54 to 82 million. In this scenario, due to its relatively large area, the greatest proportion of the costs would be related to the re-creation of sclerophyllous vegetation.

Scenario C aims to maximize biodiversity conservation benefits, by concentrating on the restoration of ecosystems that contain a high proportion of habitat types that are the focus of measures under the Habitats Directive. This would result in a focus on degraded areas of heathland and tundra, mires, dunes and beaches and saltmarsh, and little restoration of arable, permanent crops and improved grassland. Under this scenario the total cost of **restoration** would range from **€5.16 billion to 7.87 billion,** whilst **re-creation** costs would range from **€64 to 115 million**.

Scenario D aims to maximise ecosystem services, by focusing on the restoration of ecosystems that provide the highest overall ecosystem service benefits. This would result in a relatively even spread of restoration with all ecosystems restored, though with a focus on forests, and to a lesser extent semi-natural grasslands, heathland and tundra, and mires. The costs of **restoration** under this scenario would range from **€5.87 billion to 8.59 billion**, with **re-creation** costs ranging from **€69 to 129 million**.

Scenario E attempts to provide a balanced set of ecosystem restoration targets that takes into account the biodiversity conservation importance of ecosystems (as assessed under Scenario C) and ecosystem services (as assessed for Scenario D). Under this scenario the percentage allocation is relatively uniform, although it focusses on heathland and tundra and mires, and to a lesser extent all other ecosystems, other than arable, permanent crops, improved grassland and sclerophyllous vegetation, which only have a 10% target. The costs of **restoration** under this scenario would range from **€6.20 billion to 8.97 billion**, with a distribution amongst ecosystem types that is very similar to that under Scenario D. **Re-creation costs** and their distribution under **Scenario E** are also similar to Scenario D, ranging from **€66 to 122 million**.

When interpreting these results it should be borne in mind that there are limitations that need to be taken into account. Firstly, the minimum and maximum cost estimates are often highly divergent, sometimes differing by an order of magnitude. This is because the minimum and maximum values purely reflect the different estimates of degradation, and these are highly uncertain for many ecosystems, especially highly modified ecosystems where degradation is difficult to define and little monitored.

The estimated individual Member State costs (as presented in the main report, rather than here) should also be treated with particular caution as they are based on overall EU ecosystem degradation and conversion rates. Actual degradation and conversion rates will vary amongst countries, and therefore costs would be lower than projected for Member States with lower than average levels of ecosystem degradation, and higher if degradation levels are above average.

Furthermore, the cost estimates of meeting Target 2 are not able to take into account changes in demand and supply of ecosystem management measures, and instead assume average fixed rates (as for example obtained from agri-environment costs). As a result the costs of some actions that may be required over a high proportion of the land may be underestimated. Most importantly the costs of meeting Target 2 with regard to the maintenance of ecosystems is almost certainly significantly underestimated in this study because Target 2 requires maintenance over the entire land area. Thus very much higher payment rates than the current averages used in this study would probably be required to even approach the target, let alone achieve it. Consequently it is unrealistic to assume that the full maintenance of all areas of each ecosystem will be achieved.

A further constraint on this study has been that the maintenance cost estimates have to assume that the allocation of funds is in accordance with the theoretical need. For example payments to avert agricultural abandonment match the annual rate of abandonment. However, in reality, more wide-ranging payments covering areas at possible risk will need to be made to achieve the policy objective. But this additional required coverage cannot be estimated reliably. Therefore the estimated costs of maintenance should be treated as a theoretical minima.

Lastly it is important to emphasise that the cost estimates assume that measures included in the reference scenario will be fully implemented, and for some this will require stronger regulation and enforcement than has often occurred to date. If such regulations, such as those relating to CAP cross-compliance standards, pollution controls and measures under the WFD are not implemented as intended under the instruments then either Target 2 will not be achieved or additional funding will be required beyond that estimated here to implement alternative measures.

In conclusion, despite all the limitations of the study, it is obvious that although many existing measures and funding instruments are contributing to the maintenance, restoration and re-creation of ecosystems in the EU, a substantial increase in funding will be required to achieve Target 2 if the principal approach used is an expansion of incentive measures. In this respect the results are also consistent with the conclusions of other recent studies, although it is difficult to compare them directly. But, it is important to note that the costs of achieving Target 2 could be reduced by adopting new and more ambitious EU level regulations, for example expanding the range of mandatory cross-compliance standards. Member States might also be able to reduce costs further by establishing regulations beyond areas of EU competency (for example relating to spatial planning and forest management measures).

Although the additional funding needs are substantial it is important to consider their benefits, and to put them into context regarding available EU and national funds.

The potential benefits of achieving Target 2

There is now a wide appreciation of the economic and social values of ecosystems through the services and end benefits that they provide. Indeed this is explicitly recognised in the Biodiversity Strategy and other EU policy statements, such as the Resource Efficiency Roadmap and the Green Infrastructure Strategy. It can therefore be confidently anticipated that the maintenance and restoration of ecosystems will provide substantial benefits from the increased supply of ecosystem services that are dependent on healthy sustainably managed ecosystems (such as those related to water provisioning, carbon sequestration and storage, flood management, coastal protection, fisheries, cultural values, human health and recreation).

Unfortunately it is very difficult to quantify cost benefit ratios as published scientific studies are relatively sparse and patchy. Nevertheless, international evidence suggests that restoration benefits tend to outweigh costs. Furthermore, a few European examples reviewed in this study relating to the restoration of peatlands, riverine forests, rivers, coastal saltmarshes and marine ecosystems (through the establishment of marine protected areas) show that restoration can provide substantial benefits that can greatly outweigh costs.

The potential options for financing the achievement of Target 2

This study provides an overall review of potential financing instruments for the achievement of Target 2, and focuses further on the potential for using private financing instruments. This reveals that public funding is almost certainly going to remain the main source of funding measures to maintain and restore ecosystem to 2020 at the very least. This is in part because innovative private financing instruments are still in their early stages of development and testing, but more fundamentally the ‘public goods’ characteristic of biodiversity resources (ie their public but non-exclusive benefit) means that it is not possible for an investor to reap their full return on investment. Therefore, there is a clear justification for the use of public funds in supporting many ecosystem services directly, or leveraging private sector investment.

Currently a wide range of public funding instruments may contribute to the achievement of Target 2, of which the most important at an EU-scale is by far the CAP (with suggestions that €100 billion may be available for environmental purposes over the 2014-2020 programming period), although there are also significant and increasing opportunities to support ecosystem restoration under the Structural and Cohesion Funds. Funding under the LIFE programme also makes a very important contribution to the maintenance and restoration of ecosystems especially with the Natura 200 network. Its principal value is its ability to fund the development, testing and demonstration of innovative conservation and restoration techniques. However, it is widely acknowledged that the volume of the LIFE budget is insufficient in relation to the need for further biodiversity conservation and restoration actions.

It is clear from this study that CAP funds, in particular those under Pillar 2, already make major contributions to the management and restoration of ecosystems. Moreover, they have the potential to make a further vital contribution to achievement of Target 2. In fact the 2013 reforms include changes to rural development objectives that may provide increased opportunities to contribute to ecosystem restoration priorities. However, a more certain detrimental impact of the reforms and recent agreement on the EU’s Multi-annual Financial Framework (MFF) is the agreed cut to the CAP budget, which will result in a greater reduction in Pillar 2 funds. This will have significant implications for those rural development measures with the greatest potential to support restoration priorities. Furthermore, these cuts are likely to be exacerbated by the new measure that allows a selection of Member

States to transfer funds from Pillar 2 to Pillar 1 funding. Pillar 1 greening measures, including Ecological Focus Areas (EFAs) will make a much lower contribution to the achievement of Target 2 because large areas of land will be exempt and EFA rules will now allow some land uses with in them that will provide little ecological benefit. Consequently, although Pillar 1 measures should be implemented as effectively as possible, the priority as far as the achievement of Target 2 is concerned is to maintain Pillar 2 funding as much as possible and ensure it is targeted towards ecosystem maintenance and restoration priorities.

Although the most significant funding sources for biodiversity measures continue to be public (national and European) funds there is the potential to increase private financing through a variety of mechanism, especially when combined with public funds. The following three mechanisms were identified as having particular potential in the near future: payments for ecosystem services (PES), product labelling and certification, and bio-carbon markets. All directly impact on the ecosystem (ie finance changes on the ground, rather than acting as a supporting framework), are well understood - in some cases are already happening - and have potential to have large-scale influences. PES and labelling work well for maintenance measures while bio-carbon markets have the potential to support these measures where the carbon markets recognise future carbon emissions avoided. Both PES and carbon markets have a high potential for funding restoration measures.

Steps that could be taken to further encourage and support private financing of biodiversity maintenance and restoration measures (and the related delivery of ecosystem services) include using public funds (most obviously CAP agri-environment funds) to leverage private sector investment (eg by sharing the risk of investment and helping to raise capital), shaping subsidies to better support ecosystem maintenance and restoration, providing tax incentives for ecosystem restoration, expanding markets for products associated with restored ecosystems (eg by promoting product labelling), supporting pilot or demonstration projects and increasing the capacity of governmental authorities, NGOs and foundations to broker and organise private investment. In addition, further research is required to support initiatives that aim to increase private sector investment, particularly through innovative financial mechanisms.

Overall conclusions

There are a wide range of EU and national measures, including regulations and funding for practical ecosystem management and restoration that are already contributing to the objectives of Target 2. Most notably, if properly implemented, the WFD will maintain aquatic ecosystems and result in sufficient restoration to meet most aspects of Target 2 requirements. However, for all other ecosystems significant additional practical maintenance and restoration measures are clearly required to achieve Target 2.

Whilst it appears appropriate to achieve some ecosystem maintenance and restoration through new or strengthened regulations (especially regarding air pollution), most will require proactive actions encouraged through incentive payments that will therefore need significant additional funding. Some of the required funding may be obtainable from the private sector through new innovative financing mechanisms, such as PES schemes. However, it seems inevitable that up to 2020 most of the funding will need to be provided by public EU funds and national contributions. In this respect the CAP has a key role to play, especially through the adequate funding of agri-environment and similar measures that can contribute to Target 2. However, other large EU funds, especially those linked to Cohesion Policy, also have the potential to make major contributions to the achievement of the Target.

Although the costs of achieving Target 2 are significant, it is important to bear in mind the substantial benefits that would arise from the achievement of the target, and the Biodiversity Strategy as a whole. Whilst current data are insufficient to quantify exact cost-benefit ratios, there is good evidence to show that investing in the maintenance and restoration of ecosystems and their services will provide substantial environmental, social and economic gains.

1 INTRODUCTION

1.1 Background

In March 2010, the European Council adopted the new EU target to 'halt biodiversity and ecosystem service loss by 2020, to restore ecosystems in so far as is feasible, and to step up the EU contribution to averting global biodiversity loss'. This target explicitly recognises the importance of ecosystem services provided by biodiversity in addition to the need to protect biodiversity for its intrinsic value. A longer term vision was also adopted: 'By 2050, EU biodiversity and the ecosystem services it provides – its natural capital – are protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided.' To help achieve the 2020 target, the Council called on the Commission to develop, in cooperation with Member States, an EU post-2010 Biodiversity Strategy, including sub-targets and feasible and cost-effective measures and actions needed to achieve them.

A further driver for biodiversity conservation and restoration in the EU is its commitments to the Convention on Biological Diversity (CBD) and the outcome of the CBD's 10th meeting of the Conference of the Parties (held in Nagoya in October 2010), which adopted a new global Strategic Plan for biodiversity. This includes a vision that by 2050 'biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people'. To implement this vision, the Plan sets out 20 headline targets under five strategic goals to be achieved by 2020. Consequently, the new EU 2020 Biodiversity Strategy[4] (European Commission, 2011a), adopted on 3 May 2011 and endorsed by the Environment Council on 21 June 2011 has a dual purpose: to support the EU and global 2020 biodiversity targets.

To achieve its overall 2020 target the EU Biodiversity Strategy includes six interrelated subtargets (see Box 1.1). The focus of this study, is Target 2 which is that "By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure[5] and restoring at least 15% of degraded ecosystems". Target 2 is also supported by three key actions, which aim to improve knowledge of ecosystems and their status, develop a green infrastructure for the EU and ensure no-net-loss of biodiversity (Box 1.2).

Target 2 aims to support Target 15 of the CBD's 2020 Strategic Plan, which states that 'By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least

[4] Hereafter referred to as the EU Biodiversity Strategy

[5] Green infrastructure is defined by the European Commission in its 2013 Green Infrastructure Strategy (COM(2013) 249 final as "*a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, GI is present in rural and urban settings*."

15% of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification'. In addition, it should contribute to ensuring 'ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and wellbeing, are restored and safeguarded (Global Target 14).

Box 1.1 The EU 2020 Biodiversity Strategy targets

Target 1: To halt the deterioration in the status of all species and habitats covered by EU nature legislation and achieve a significant and measurable improvement in their status so that, by 2020, compared to current assessments: (i) 100% more habitat assessments and 50% more species assessments under the Habitats Directive show an improved conservation status; and (ii) 50% more species assessments under the Birds Directive show a secure or improved status.

Target 2: By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems.

Target 3
A) Agriculture: By 2020, maximise areas under agriculture across grasslands, arable land and permanent crops that are covered by biodiversity-related measures under the CAP so as to ensure the conservation of biodiversity and to bring about a measurable improvement in the conservation status of species and habitats that depend on or are affected by agriculture and in the provision of ecosystem services as compared to the EU2010 Baseline, thus contributing to enhance sustainable management.

B) Forests: By 2020, Forest Management Plans or equivalent instruments, in line with Sustainable Forest Management (SFM)21, are in place for all forests that are publicly owned and for forest holdings above a certain size (to be defined by the Member States or regions and communicated in their Rural Development Programmes) that receive funding under the EU Rural Development Policy so as to bring about a measurable improvement in the conservation status of species and habitats that depend on or are affected by forestry and in the provision of related ecosystem services as compared to the EU 2010 Baseline.

Target 4: Fisheries: Achieve Maximum Sustainable Yield (MSY) by 2015. Achieve a population age and size distribution indicative of a healthy stock, through fisheries management with no significant adverse impacts on other stocks, species and ecosystems, in support of achieving Good Environmental Status by 2020, as required under the Marine Strategy Framework Directive.

Target 5: By 2020, Invasive Alien Species and their pathways are identified and prioritised, priority species are controlled or eradicated, and pathways are managed to prevent the introduction and establishment of new invasive alien species.

Target 6: By 2020, the EU has stepped up its contribution to averting global biodiversity loss.

Box 1.2. Actions to support Target 2 on ecosystem maintenance and restoration

Action 5: Improve knowledge of ecosystems and their services in the EU
5) Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020.

Action 6: Set priorities to restore and promote the use of green infrastructure
6a) By 2014, Member States, with the assistance of the Commission, will develop a strategic framework to set priorities for ecosystem restoration at sub-national, national and EU level.
6b) The Commission will develop a Green Infrastructure Strategy by 2012 to promote the deployment of green infrastructure in the EU in urban and rural areas, including through incentives to encourage up-front investments in green infrastructure projects and the maintenance of ecosystem services, for example through better targeted use of EU funding streams and Public Private Partnerships.

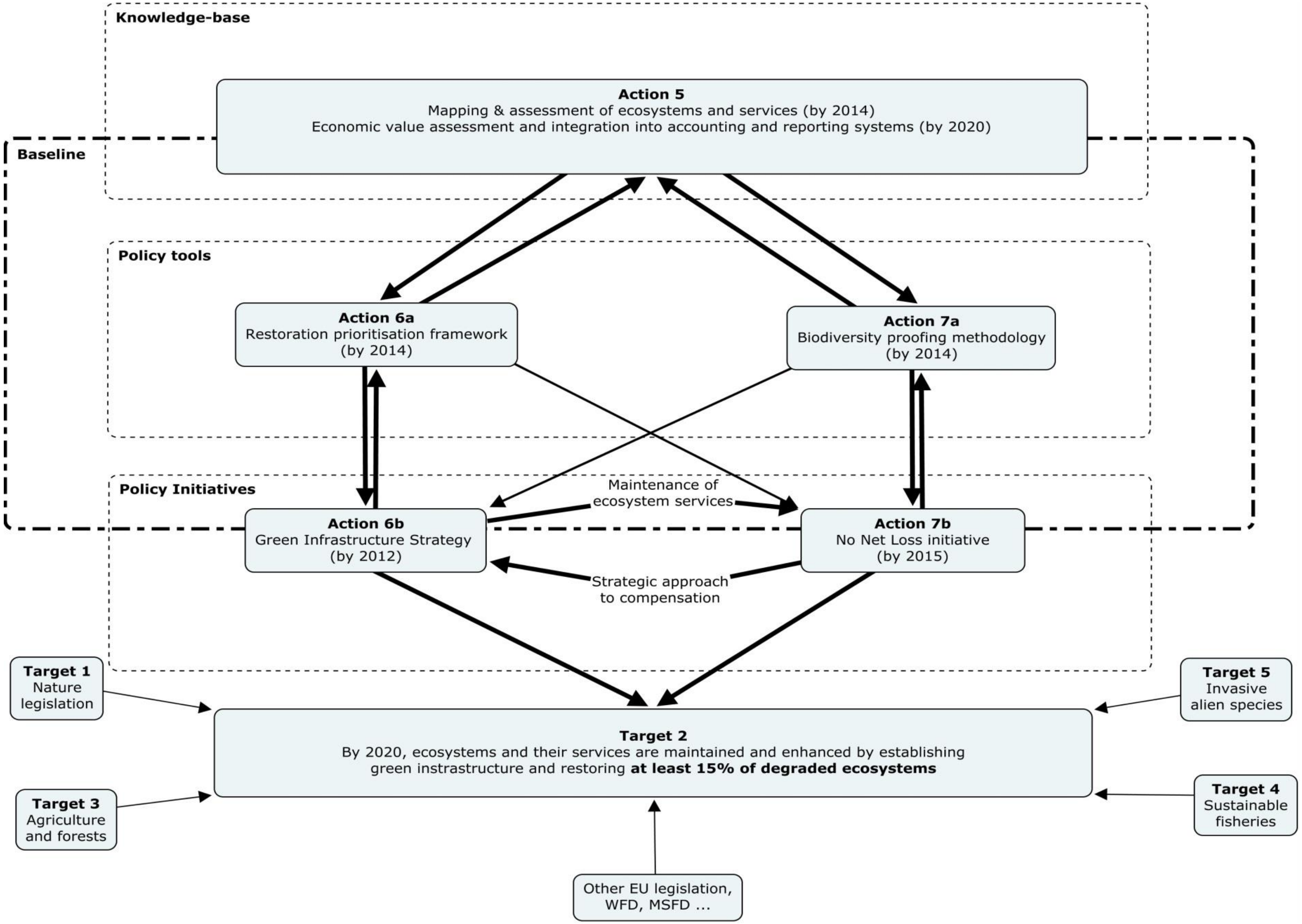
Action 7: Ensure no net loss of biodiversity and ecosystem services
7a) In collaboration with the Member States, the Commission will develop a methodology for assessing the impact of EU funded projects, plans and programmes on biodiversity by 2014.
7b) The Commission will carry out further work with a view to proposing by 2015 an initiative to ensure there is no net loss of ecosystems and their services (eg through compensation or offsetting schemes).

Target 2 of the EU's Biodiversity Strategy is an ambitious target because it relates to all ecosystems and their services, and the whole environment and not just protected areas. It therefore extends the actions envisaged to maintain and restore Favourable Conservation Status of threatened species and habitats of Community interest[6] under Target 1. Target 1 actions will primarily need to focus on the Natura 2000 network, but actions in the wider environment, through for example green infrastructure enhancement, will also be needed to support the network's connectivity and coherence. Consequently Target 1 and 2 are mutually dependent: Target 1 measures will contribute to the achievement of Target 2, whilst the achievement of Target 2 will be necessary for the achievement of Target 1. Targets 3, 4 and 5 to some extent support and overlap with Target 2.

There are also interactions between the various actions that support Target 2, and these and the links to other targets are summarised in Table 1-1 below.

[6] Formally these are species and habitats listed in Annexes I and II and/or IV or V of the Habitats Directive. For the purposes of this study we also include species listed in Annex I of the Birds Directive.

Table 1-1 Biodiversity Strategy to 2020 – Linkages within Target 2 and with other targets



Target 2 is also ambitious because it goes beyond the maintenance of ecosystems, many of which are under on-going significant pressures, to the restoration of at least 15% of degraded ecosystems. According to the European Commission's Biodiversity Strategy Impact Assessment (European Commission, 2011b), habitat restoration is "*Actively assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed, although natural regeneration may suffice in cases of low degradation. The objective should be the return of an ecosystem to its original community structure, natural complement of species, and natural functions to ensure the continued provision of services in the long term*." Although the interpretation of this definition is complex it is clear the desire is not merely to restore an ecosystem's key species and ecological processes. Instead, the aim is full restoration of structure, species community and ecological processes and functions. However, as discussed further in section 3.2 this level of restoration would be prohibitively expensive in most situations and often impossible.

It is also important to note that Target 2 explicitly mentions the need to restore ecosystems for ecosystem services (as well as for intrinsic nature conservation values) as the multiple benefits of ecosystems must be maintained. Furthermore, although a range of approaches may be used, it is implied that green infrastructure is the primary means of achieving the target, because it is considered that the spatial integration of measures in the landscape and the restoration of connectivity is essential. Restoration measures will therefore need to be spatially planned and implemented strategically to ensure their efficiency and coherence. A recent study for DG Environment (Mazza et al, 2012) revealed that many current EU and national policy instruments exist that can support the development of green infrastructure. But more may be achievable with new policy instruments, which may require further political enrolment as part of the EU's anticipated Green Infrastructure Strategy (see Action 6b in Box 1.2).

The ambition of Target 2 in terms of the extent of restoration is also highly dependent on the proportion of ecosystems that are considered to be degraded. Although, as discussed in section 3.2.3, there is much uncertainty over the definition of degradation and actual degradation levels, there is strong evidence from the European Environment Agency's (EEA) recent Biodiversity Baseline assessment (2010a; EEA, 2010b), and other sources, of widespread detrimental impacts of human activities on ecosystems in the EU. Consequently, the EU Biodiversity Strategy states that "In the EU, many ecosystems and their services have been degraded, largely as a result of land fragmentation. Nearly 30% of the EU territory is moderately to very highly fragmented". The scale of restoration actions will therefore clearly be substantial.

It is also important to note the Commissions' Biodiversity Strategy Impact Assessment indicates that the target is indicative, and the level of ambition should be reviewed in 2014. But it is unlikely that the target will be reduced, as it is the minimum level needed to comply with CBD obligations. Indeed, the Impact Assessment indicates that a higher figure (for example 30%) may be required to achieve the EU's more ambitious overall 2020 headline target and 2050 vision, especially bearing in mind the level of ecosystem fragmentation in the EU. Furthermore, although the current 15% target is in some respects ambitious, the EU has many policy and financial instruments that are already in place and contributing to ecosystem conservation and restoration. Thus the potential for restoration should also take into account existing measures that are likely to indirectly support the target, including measures for the implementation of the Birds and Habitats Directive (for Target 1) including LIFE+ funded projects, as well as other funding, in particular the Common Agricultural Policy (CAP) (especially agri-environment measures), Cohesion Policy and the Common Fisheries Policy (CFP). Restoration actions expected under the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD) should also result in significant improvements in the condition of ecosystems in future. Relevant research activities are also financed through the Framework Programmes for Research and Technical Development. There is also growing recognition of the potential for private investment, such as through payments for ecosystem services (PES schemes) and habitat banking (in support of Action 7b) to restore degraded ecosystems.

Perhaps most importantly, there is evidence (Aronson et al, 2010; Bullock et al, 2011; de Groot, 2008; Nellemann and Corcoran, 2010; Russi et al, 2013) that the costs of many restoration actions will be more than compensated for by their benefits, such as stimulating green growth and territorial cohesion, facilitating ecosystem-based climate change mitigation and adaptation and the enhancement of other ecosystem services. Green infrastructure measures may also result in cost-savings compared to more conventional engineered measures (eg coastal habitat restoration compared to hard sea defences). But as discussed in section 3.4.3, information on the likely costs and benefits of ecosystem maintenance and restoration in the EU is scattered and difficult to obtain. Therefore the Impact Assessment concludes that there is “currently not sufficient evidence of how much restoration would take place under existing EU policy, whether additional efforts would be needed to reach 30%, and what their costs and benefits would be, to take a fully informed decision. Therefore, the chosen level is initially the minimum compliance with respect to international commitments.”

By collating data and assessing the additional costs of achieving Target 2 (ie beyond the costs of existing and anticipated measures that will contribute to ecosystem maintenance and restoration by 2020), this study aims to inform debates on the potential strategic options for achieving the target effectively and efficiently.

Box 1.3 Key definitions

Biodiversity: The variability among living organisms from all sources including terrestrial (above and below ground), marine and other aquatic ecosystems and the ecological complexes of which they are part. This concept covers the diversity of genes, species and ecosystems. Based on CBD

Connectivity: the extent to which ecosystems and natural areas are linked together in fragmented landscapes.

Ecosystem: A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit. In this study we use the term to refer to the broad divisions used to calculate Target 2 costs (cf habitats below).

Ecosystem services: The direct and indirect contributions of ecosystems to human wellbeing. They can be categorised in four main types: provisioning services (eg food, water, fuel); regulating services (eg flood and disease control); supporting/habitat services (eg nutrient cycling); and cultural services (eg recreation).

Habitat: In this study we primarily use this term to refer to habitat types listed in Annex I of the Habitats Directive (which are also known as biotopes).

Fragmentation: the division of an ecosystem or habitat into distinct parts. Fragmentation can result from infrastructure development, such as roads and railways, or natural occurrences like forest fires.

Green infrastructure: According to a recent study for the Commission (Naumann et al, 2011), "Green infrastructure is the network of natural and semi-natural areas, features and green spaces in rural and urban, terrestrial, freshwater, coastal and marine areas, which together enhance ecosystem health and resilience, contribute to biodiversity conservation and benefit human populations through the maintenance and enhancement of ecosystem services. Green infrastructure can be strengthened through strategic and co-ordinated initiatives that focus on maintaining, restoring, improving and connecting existing areas and features as well as creating new areas and features."

Pressures: Habitat loss, overexploitation of natural resources, the introduction and spread of invasive species, pollution and climate change are the five key pressures on biodiversity.

Resilience: The ability of an ecosystem to buffer and adapt to changes as well as recover after being disturbed.

Re-creation: restoration where the ecosystem has been destroyed in the restoration site

Restoration: the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SERI, 2004) – see further discussion in section 3.2.2.

1.2 The study's objectives

The overall objective of this study, as stated in the project specification is

> "a scoping study on the financing needs to implement Target 2 of the EU 2020 Biodiversity Strategy. It is expected to compile existing information and undertake further analysis to provide an estimation of financing needs for implementing Target 2 of the EU 2020 Biodiversity Strategy focusing in particular on the restoration of at least 15% of degraded ecosystems, based on the Actions outlined in the Strategy to reach this Target. It should also provide a general analysis of the main financing means that will need to be considered, looking in particular at the extent to which private sources could complement financing from the main EU instruments."

The study's tasks, adapted from the project specification, are described below.

Task 1: Compilation of existing information

The study will compile existing information, from DG Environment commissioned studies, and other sources (eg LIFE projects) on:

1. Costs and benefits of restoration of ecosystems, including the establishment of green infrastructure, focussing on EU experiences, but drawing on wider data where it is necessary to fill gaps. The main focus of the work should be on the costs, in order to provide a range of restoration costs for different ecosystems, which should include both investment and maintenance costs.

2. Potential financing instruments for restoration and green infrastructure.

Task 2: Developing possible general scenarios for implementation

This will provide an assessment, as a reference point, of the current extent of degradation of ecosystems and ecosystem services, based on the EU 2010 Biodiversity Baseline, and other relevant sources. This reference scenario will take into account forthcoming ecosystem conservation and restoration measures expected under existing legislation, including the Habitats and Birds Directives, the WFD and the MSFD.

A number of potential key scenarios for implementing the target of achieving at least 15% of degraded ecosystems in the EU, will be explored, including *inter alia*:

- achieving at least 15% of restoration for each main type of ecosystem as described in the EU 2010 Biodiversity Baseline;
- achieving at least 15% of restoration at the level of each EU Member State, or
- achieving at least 15% at EU level overall, with some flexibility on the contribution from each Member State (or through a 'burden sharing' scheme), or on the different types of ecosystem covered.

Under each scenario, restoration will be guided by a prioritisation framework using criteria to be developed as part of this task, such as the extent of degradation of ecosystems; provision of key ecosystem services and cost-benefit ratios of restoration.

Task 3: Producing cost estimates for the implementation of Target 2

This task will use the scenarios developed in Task 2 as a basis for estimations of the potential costs of implementation of Target 2 of the EU 2020 Biodiversity Strategy. The estimation will focus on the scenario of a fixed allocation of 15% of ecosystems for each type of ecosystem, and explore how these might vary if there is more flexibility in the target, within the constraints of a prioritisation framework.

The approach will enable the scaling up of project-based restoration costs and allow a general sensitivity analysis to be carried out on how the assumptions made to scale up the costs affect the estimates, and the possible implications of shifting the focus of restoration to ecosystem services.

The task will also explore the extent to which the elements of costs and benefits of no-net-loss policies or projects reviewed under Task 1 could be extrapolated to produce broad-brush estimates of implementing no-net-loss approaches in the EU.

Task 4: Exploring potential financing means to reach Target 2

Building on the review of potential instruments performed under Task 1, the task will provide an analysis of the potential contribution that financial instruments could make to achieving the target of 15% of restoration. A particular focus will be given to innovative sources of funding, and funding from the private sector.

The task will also examine a small number of examples of practical implementation at national or sub-national level focusing, for example, on payments for ecosystem services from private sources (including through public private partnerships) or support to businesses which could invest in green infrastructure projects. The conditions under which such schemes could take place will be explored.

The volume of potential financial sources will be examined in the existing institutional and regulatory context, and the means by which it could be scaled up through further action at local, national or EU levels will be explored.

1.3 Outline of the structure and contents of this report

Although most of this study has been structured around the tasks outlined above, this report has reorganised the methodological descriptions of the work, and assembled information and results to improve its coherence and readability. The following paragraphs therefore outline the contents of each of the chapters and the tasks that they address.

As further background (and as part of Task 1.1) chapter 2 briefly reviews the methods and results of previous studies that have attempted to calculate the costs and benefits of ecosystem maintenance and restoration in relation to various environmental objectives, such as the management of Natura 2000 sites, and restoration of water and soil related ecosystem services. These estimates are later referred to in chapter 10, which compares the results of this study with previous estimates.

Chapter 3 provides a detailed explanation of the methods that have been used in the study to calculate the costs of maintaining each ecosystem and restoring 15% of their degraded area (ie with respect to Tasks 1.1, 2 and 3). The chapter includes further discussion of meaning of the EU target and definitions of restoration and degradation.

The review of the costs of ecosystem maintenance and restoration carried out under Task 1.2 is documented in chapters 4-6. The most important costs in terms of the additional costs of achieving Target 2 are likely to be those associated with practical ecosystem-specific maintenance and restoration measures, and therefore these have been examined in detail in this study (see chapter 6 below). However, to put these into context with respect to the specified supporting actions for Target 2, chapter 4 firstly provides an analysis of the likely costs of improving knowledge on ecosystems (EU 2020 Biodiversity Strategy Action 5), promoting a green infrastructure (Action 6) and ensuring no-net loss of biodiversity (Action 7). To avoid double counting of some costs this analysis does not deal with practical measures that directly result in the maintenance or restoration of habitats (which are covered in chapter 6). Instead it focuses on strategic research, policy development and planning activities etc that will support practical habitat measures.

Some of the measures required to maintain and restore habitats are generic wide-scale pressures (such as nitrogen deposition) that are best dealt with by tackling their source, although in some cases in-habitat mitigation measures may also be taken (eg increased grazing of grasslands that are suffering from nutrient enrichment). These generic pressures and corresponding measures are dealt with in chapter 5, before consideration of further habitat-specific measures. It is difficult to estimate the level of actions and costs that would be necessary to meet the 15% target for these measures, but data on these issues are reviewed where available in the published literature.

Chapter 6 provides a detailed assessment of the potential costs of maintaining and restoring each main type of ecosystem recognised in this study. It therefore firstly identifies the key pressures that affect each ecosystem (as part of Task 2) and then the key measures that can be taken to alleviate them in order to maintain and restore each ecosystem. Baseline degradation levels and pressures are quantified for each ecosystem and then 2020 levels projected according to a reference scenario which takes into account foreseen measures and policies. The assessment of pressures draws on published literature, but also the results of monitoring data collected by Member States on habitats of Community interest in accordance with Article 17 of the Habitats Directive.

Chapter 7 addresses Task 3 and provides an estimate of the total costs of achieving the overall EU target of maintaining and restoring 15% of degraded ecosystems. These costs are calculated for the EU as a whole and for each Member State. To achieve this a number of scenarios (developed under Task 2) are investigated for the disaggregation of the overall EU target, comprising allocations that result in 15% restoration for each ecosystem, minimise the costs of restoration, maximise nature conservation benefits, maximise ecosystem service benefits and maximise a mix of nature conservation and ecosystem service benefits.

Chapters 8 and 9 address Task 4. Firstly, chapter 8 provides an overall review of potential financing instruments for ecosystem maintenance and restoration, and the implementation of green infrastructure objectives. Chapter 9 then examines in further detail the potential for using private financing instruments to support the maintenance and restoration of ecosystems and achievement of the EU target.

Lastly, chapter 10 compares the results of the study to previous relevant studies and discusses their accuracy, reliability and limitations. Based on this it then identifies data and research gaps that would need to addressed in order to provide more comprehensive, precise and robust estimates of the costs of achieving Target 2 of the Biodiversity Strategy.
The chapter also briefly outlines further work that could be carried out to increase the availability of funding for ecosystem maintenance and restoration, in particular through the use of innovative private financing instruments.

2 PREVIOUS STUDIES OF THE COSTS AND BENEFITS OF MAINTAINING AND RESTORING ECOSYSTEMS

2.1 Overview and cost types

A number of studies have been carried out on the costs of achieving a variety of environmental objectives in the EU, which are of relevance to the maintenance and restoration of ecosystems, and the establishment of green infrastructure. Most are studies that have attempted to estimate the costs of meeting various environmental targets through rural land management, including the management of agricultural land as well as forested areas. These range from fairly detailed estimates in relation to specific objectives and areas, such as the costs for achieving Favourable Conservation Status on Natura 2000 sites (which is most relevant to Biodiversity Strategy Target 1, but will support Target 2) to more generic costs associated with maintaining High Nature Value (HNV) farming across the farmed landscape, addressing soil erosion and water resource and quality issues. Some costs have been estimated for the EU-27, while other exercises have been undertaken at the individual Member State level. Most studies focus on the costs associated with delivering specific environmental needs, with the majority of studies focusing on the costs of managing the land to meet biodiversity targets and to address soil degradation.

In some cases the studies have attempted to partition costs according to whether they are direct realised costs (eg for design, planning, physical works, and administration etc) or opportunity costs, which may be compensated for by for example agri-environment payments. This approach was for example taken in a recent study for DG environment of opportunity costs (Kaphengst et al, 2010), as outlined in **Error! Reference source not found.**. However, the separation of costs into such categories is complex and the quantification of opportunity costs is a very difficult issue, unless they have been calculated for compensation payments. Most of the studies below do not therefore separate out cost types, and uncompensated opportunity costs are not normally calculated and included in cost assessments.

Table 2-1 A cost typology for biodiversity action

Source: Kaphengst et al (2010)

Cost category	Types of Costs	Examples
Financial Costs	**Costs of resources expended:** Costs of capital, labour, materials, energy Capital costs and recurrent management costs Administrative and transaction costs involving financial outlay	Labour and materials for fences around nature reserves Salaries and equipment of biodiversity researchers Materials, labour and equipment for construction of visitor centres Costs of developing and administering species action plans
	Costs that reflect opportunity costs: Payments to compensate for income foregone Compensation payments for foregone development/ exploitation rights Land purchase (reflecting income from land in alternative use)	Agri-environment payments to compensate for loss of cereals output from leaving fallow land for nesting birds Compensation payments to fishermen for establishment of marine nature reserve Cost of purchase of farmland to establish new wetland reserve
Wider Economic Costs	**Uncompensated opportunity costs:** Lost income from foregone development Loss of socio-economic opportunities Output restrictions on exploitation of natural resources	Loss of income from prevented commercial and industrial development Foregone opportunities for job creation and cohesion Loss of output of fisheries, wood, minerals, energy etc.

2.2 Costs of biodiversity conservation within the Natura 2000 network

Several studies have sought to estimate the costs associated with the management needs of the Natura 2000 network, mainly on the basis of data provided by Member States', which vary considerably in their estimation methods. A first estimate of the **costs of managing Natura 2000 sites across the EU-25 Member States** was produced in 2004, based on the Markland Report (Markland, 2002), which estimated that €6.1 billion per year was needed[7]. This estimate has recently been reviewed and revised within the frame of a project undertaken by IEEP, Ecologic and GHK for the European Commission (Gantioler *et al,* 2010). The study assessed data supplied by 25 Member States and produced a **revised estimate of 5.8 billion €/yr based on an extrapolation to the EU-27 of the average cost of 63 €/ha/yr**. The revised estimates are considered to be more reliable in some respects because, for example, estimates of the areas requiring management were improved and a more

[7] COM 2004(431)

consistent methodology for estimating costs was used. However, it is recognised by IEEP and others that the revised estimate is likely to be a significant underestimate of the total costs needed to bring Natura 2000 sites into Favourable Conservation Status (which relates to Target 1). The Commission's earlier figures suggested an annual cost of around **107 €/ha** and a BirdLife International (2009) report indicated costs associated with management of the Natura network at **128 €/ha/yr**, based on estimates from six Member States provided by the organisation's national partners.

The underestimation of costs is considered to be due to variations in the interpretation of conservation objectives and needs, and variations in data. In some cases the Member State estimates were based on the costs of maintaining Natura 2000 sites in their current condition and in only a few cases were they based on the costs of achieving Favourable Conservation Status of habitats and species of Community interest within them (which is the aim of the Habitats Directive). Very often estimates were based on what is achievable under currently available budgets and in some situations the cost estimates only included management that is additional to that required by law (UK for example). Only a few Member States estimated the expenditure that would be needed ideally if the resources were available. Only Spain provided two estimates, the cost of managing the network within existing resources and another estimate of what would be 'desirably' spent if resources were available, the latter being 60% higher. A key reason for the likely underestimate of the costs is therefore the fact that many countries focused on historic and/or budgeted investment costs, and only a few provided information on future needs.

2.3 Costs of biodiversity conservation in the wider environment

The EU studies and some national studies, such as in Germany (Güthler and Oppermann, 2005), provide some insights and useful data that may help with the calculation of wider ecosystem maintenance and restoration costs. However, Natura costs are unlikely to be representative of costs in the wider environment because, amongst other things, they are likely to differ in their habitat management needs (being highly specific and biodiversity focussed), vulnerability to pressures and costs of action (eg due to low profitability and low land prices). Estimating the costs of environmental management in the wider environment will therefore be especially important for this study.

HNV farming and forestry systems are commonly found in Natura 2000 sites, but they are much more widely distributed and with less protection a large proportion of them of have been degraded in recent years due to abandonment or intensification and fragmentation. Therefore widespread actions are likely to be needed to maintain and restore them, and consequently the costs of this are likely to be a significant proportion of the costs of meeting Target 2 in terrestrial ecosystems. Two estimates have been produced on the scale of support needed to maintain HNV farming practices in the EU-27.

Beaufoy and Marsden (2010) provide costs for the introduction of a targeted scheme for HNV farming under Pillar One of the CAP, as part of a wider strategy for maintaining HNV farming in the EU-27. Rough calculations suggest that, **to maintain HNV farming systems in all Member States would require expenditure of 16 billion €/yr, assuming an average payment for HNV farming of 200 €/ha/yr** over an estimated HNV farmland area of 80 million hectares (likely to be a significant overestimate of the actual HNV farmland area). This cost estimate, however, is only one element of the total potential funding needed to maintain HNV farming. On top of this cost would also be costs associated with more specific and targeted management needs, for example for certain threatened species or habitats, funded for example through the agri-environment measure, as well as costs associated with capital investments, and presumably also Less Favoured Area Payments (LFA) type payments, although this is not made clear.

A study by Kaphengst et al (2010) attempted to estimate the total economic costs associated with maintaining HNV farming through the agri-environment measure in the EU-27. To do this, an average payment rate for HNV management was calculated, based on data on a range of relevant management practices collected from six RDPs[8] and this was applied to an estimated target area of HNV farmland to which agri-environment actions are anticipated to be applied, again based on relevant targets identified within the RDPs and scaled up to the EU-27. An average per hectare figure for maintaining HNV grassland under the agri-environment measure (in addition to Pillar 1 direct payments and LFA payments) was derived of **169 €/ha and a total cost of maintaining HNV farming practices over 26 million hectares of HNV farmland in the EU-27 was calculated as 4.37 billion €**.

The study by Kaphengst *et al,* also attempted to calculate the costs of managing semi-natural forests (which are to some extent equivalent to HNV forest areas). Under the study, the costs were simply extrapolated from the figures derived for the costs of **management of forest Natura 2000 sites (Gantioler et al, 2010), giving a total figure for managing semi-natural forests in the EU-27 as 4.5 billion /ha (based on an average payment of 37 €/ha/yr over 150 million ha)**. However, as mentioned above Natura costs are unlikely to be representative of costs in the wider environment, so the validity of this approach is questionable. Furthermore, the figures include both privately and publically owned forests and, given that forests under public ownership are only eligible to receive funding for certain activities (establishment costs rather than management costs), the estimated costs are likely to be a significant overestimate of the costs required from the public purse.

Some national studies have provided useful insights on the costs of meeting biodiversity conservation objectives across agricultural land. These include a study by Hampicke et al (2010) of the costs of achieving ‘more natural management’ of agricultural land in Germany through the following types of management:

- Maintenance of semi-natural landscapes and extensive grassland including:
  - Grazing with sheep on neglected calcareous grasslands;
  - Grazing with suckler cows/young cattle on neglected delicate grasslands;
  - Mowing and hay production;
  - Bringing grassland that is reverting to scrub back into production through scrub removal;
- Extensification of 10% of the land under intensive grassland management;
- Protection of arable flora on low yielding arable land ;
- 7% of land to be ‘structural elements’ – includes woodland, hedgerows, strips of grassland along roads, water bodies, hedgerows etc.

[8] The six RDPs used were Austria, Bulgaria, Czech Republic, Poland, Romania, UK (England)

Estimates were produced for the total area over which the different types of management were needed and the per ha cost for each type of management was calculated, based on income foregone and additional costs. These estimates suggest that the **management practices cost 652 €/ha** and are needed over 2.3 million ha (15% of Germany's UAA) resulting in a total cost of 1.5 billion €/yr. In comparison current funding available under EAFRD for similar management in Germany is 1.25 billion €/yr.

A similar study in the Netherlands (Overmaers et al, forthcoming) estimated the costs of the conservation of meadow birds (through mosaic management and later mowing dates), other farmland birds (by avoiding pesticide use in cereal field margins, use of spring-sown cereals, creation of field margins and use of winter feed crops) and wild flora (by ecological grassland management, field margin management). Estimates suggest that in core areas 157,000 ha of land (8.6% of UAA) would need to be **managed at an average cost of 711 €/ha**, coming to a total of 111 million €/yr. If management were carried out across the farmed countryside, the area that would be needed to be under such management is estimated to be 409,000 ha (22.4% of UAA) at an **average cost of 798 €/ha**, coming to a total of 326 million €/yr. To put these figures in context, the current agri-environment budget for 2007-13 is 20.6 million € (plus the same in national co-financing).

In the UK a study by GHK (2006) was undertaken of implementing 42 Habitat Action Plans (HAPs) and Species Action Plans (SAPs) developed as part of the UK Biodiversity Action Plan process. Most of the action plans include targets for habitat maintenance and some degree of restoration. An estimate of combined annual net cost (including land purchase, administration and central costs and opportunity costs where reflected in grant rates and land purchase costs) of implementing all the HAPs was calculated by identifying appropriate per ha management, restoration and re-creation costs, and simply applying these to the HAP targets.

The estimated cost, expressed in 2005/06 prices, of achieving all the HAP targets was £321 million between 2005-2010, rising to £401 million between 2015 and 2020. The costs were found to be heavily influenced by the cost of the following five HAPs (each costing over £20 million per year), hedgerows, native woodlands, coastal and floodplain grazing marsh, blanket bog, and arable field margins. Given the widely differing ambitions of the HAPs and the difficulty of relating the habitats types to the EU ecosystem types, it is difficult to relate the conclusions from the study to the possible costs of meeting Target 2. The GHK study is therefore not explained in further detail here, but some of the habitat cost data are referred to later in the ecosystem accounts in chapter 6.

2.4 The cost of general legislative and policy measures to address invasive alien species

Invasive alien species are non-native species whose introduction and/or spread outside their natural past or present ranges pose a risk to biodiversity. Invasive alien species have been recognized as the one of the most important threats to biodiversity in Europe (EEA, 2010c), and they cause widespread and costly ecosystem degradation. Costs in lost output due to invasive aliens, health impacts of invasive aliens, and expenditure to repair invasive alien species damage is estimated to have already cost EU stakeholders at least 12.5 billion €/yr (according to documented costs) and probably over €20 billion (based on some extrapolation of costs) per year, over the past 20 years (Kettunen et al, 2009). Most of this total, ie over 9.6 billion €/yr, results from damage caused by invasive alien species to key economic sectors such as forestry, agriculture, fisheries and infrastructure (Kettunen et al, 2009). Costs caused by invasive alien species are expected to increase, as current measures are failing to control or eradicate species, and new potential invasive alien species are being introduced at an accelerating rate.

The EU Biodiversity Strategy 2020 includes Target 5 "*By 2020, Invasive Alien Species (IAS) and their pathways are identified and prioritised, priority species are controlled or eradicated, and pathways are managed to prevent the introduction and establishment of new IAS*." To achieve this target there is a need for new policy measures and cross-cutting tools at the EU level to support coordinated action to prevent the establishment of new invasive alien species, and to control existing invasive alien species, because EU and Member State legal frameworks still do not adequately address invasive alien species threats (Shine et al, 2009). Consequently, the Commission has announced that a proposal for a legislative instrument on invasive alien species will be produced in 2013.

A comprehensive dedicated EU legal framework on invasive alien species would involve the setting up of independent procedures for assessment and intervention taking into account existing national and EU legislation. EU Member States would be obliged to carry out controls at borders for invasive alien species and to exchange information on invasive alien species. It could also include mandatory monitoring and reporting procedures and efficient rapid response mechanisms.

An initial assessment of the possible level of costs associated with EU policy action on invasive alien species indicates that the implementation of a comprehensive dedicated EU legal framework on invasive alien species would cost up to 10 billion €/yr (Shine et al, 2010). This includes the costs of EU wide invasive alien species policy development, administration and coordination, risk assessments, research, stakeholder consultation and engagement, early warning system, management of key pathways of introduction, monitoring etc. This would form the obligatory baseline for Member States action on invasive alien species once the expected EU legislation on invasive alien species is in place.

2.5 Costs of soil restoration

Soil degradation is a significant problem over much of Europe Consequently, considerable work has also been carried out on the impacts of soil degradation and the costs of addressing it, for example as part of the Impact Assessment carried out for the objectives of the proposed Soil Thematic Strategy (CEC, 2006). The impact assessment estimated that the costs to society of soil degradation resulting from erosion, loss of organic matter, salinization and contamination would be up to €38 billion annually if no further actions are taken. In comparison, the annual costs of addressing soil organic matter decline for Europe have been estimated to be at least €3.4 – 5.6 billion (CEC, 2006b). However these costs did not include costs to address the ongoing degradation of soil functions and therefore the real costs are likely to exceed these values.

Building on the costs presented in the impact assessment, Kuhlman *et al*, (Kuhlman et al, 2010) assessed the costs and benefits of management measures that can address soil organic matter declines, erosion and compaction. These included residue management, conservation tillage, cover crops and the application of exogenous organic matter. The costs of the measures themselves varied considerably, but costs are also affected by the degree of soil erosion that needs to be tackled. For example, the **costs for implementing soil organic management practices in areas of low soil erosion risk range from 44 to 384 €/ha/yr, with an overall cost of 116 €/ha/yr for a measure integrating all practices**. The study estimated the potential area over which each of the measures are needed and used this as a basis for scaling-up the overall cost estimates. The total cost of addressing the loss of soil organic matter is estimated at 3.5 billion € per year based on a risk area of 30.5 million ha across Europe.

2.6 Costs of achieving water related environmental objectives

Few studies have examined the costs of delivering water quality targets through the use of public money. This may be due to the fact that, until fairly recently the main policy mechanisms used to improve water quality have been regulatory in nature through the implementation of the Nitrates Directive, the Groundwater Directive, the Sewage Sludge Directive, and more recently the Water Framework Directive (WFD). However, Member States have increasingly turned to the use of agri-environment schemes to work alongside regulation to encourage land management practices that can address issues of diffuse pollution from agriculture in particular. In some Member States, considerable investment is also made in the provision of advice to farmers and other land managers.

A summary and discussion of the results of those studies that have attempted to estimate the costs associated with the implementation of the WFD is provided in section 5.1.

2.7 Costs of maintaining and restoring marine ecosystems and fisheries

In the marine environment the main tool for ecosystem maintenance and restoration is the designation of Marine Protected Areas (MPAs). The implementation of MPAs in the European Union (EU) is driven by a number of international, EU and national obligations and initiatives to which the EU and its Member States are committed. In the case of the Common Fisheries Policy (CFP), there are opportunities for designation of MPAs for both stock recovery and habitat restoration through long term management plans aimed at achieving an ecosystem-approach to management. Although they have the potential to play a key role in *inter alia* fisheries management and conservation, their poor implementation record is particularly striking when the EU is faltering in meeting its broader targets in two key areas (eg restoring stocks to maximum sustainable yield (MSY) levels by 2015 and halting the loss of biodiversity by 2010) (Lutchman, 2007). Data on site designation contains so many limitations that drawing meaningful conclusions on extent of area designation becomes impossible[9].

[9] Numerous sites have been designated according to both the Birds and the Habitats Directives, either in their totality or partially. The data on numbers of sites and area coverage may therefore not necessarily add up. http://europa.eu.int/comm/environment/nature/nature_conservation/useful_info/barometer/index_en.htm

No assessments of costs of establishing or maintaining MPAs at the EU level have been published. However, there are various estimates at a global and national level in the UK of the costs associated with establishing and maintaining these sites for habitat restoration and conservation of marine ecosystems. Cullis-Suzuki and Pauly (2010) using various models estimate that it would cost 25-37 billion US $ annually to protect 20-30% of the global oceans. This value is higher than the estimate of cost by Balmford et al (2004) of 5-19 billion US$. At the UK level, the DEFRA Marine Bill consultation document (2006) suggested that it would cost approximately £195,000 to establish a marine protected area site and approximately £95,000 in annual running costs thereafter (Feilen, 2006).

2.8 The costs of achieving integrated environmental objectives

A limitation of most of the studies discussed above with respect the restoration of ecosystems for a broad range of services is that most studies focussed on the costs associated with meeting targets for a single rural environmental issue and did not take account of the synergies that exist in meeting other environmental objectives through the same management practices on the same area of land. However, a few studies, including two led by IEEP, have attempted to assess the costs of meeting a range of terrestrial environmental objectives through a range of integrated measures.

A UK study estimated the costs of meeting the priorities associated with the full range of rural environmental issues (Cao et al, 2009). This study provides estimates of the scale of funding needed to meet future environmental land management requirements in the UK, covering biodiversity, landscape, climate change mitigation, flood risk management, farmland historic environment, soil quality, water quality, resource protection and public access objectives associated with agricultural and forestry land uses.

The approach to the study was similar to that proposed by us for this study (see section 3.4) and followed in a recent IEEP study (see below). Firstly, the type and scale of environmental management needed to meet UK environmental policy objectives and targets was identified. Secondly, the costs of delivering the management practices identified were estimated based on current agri-environment payment rates. The continued existence of LFA payments and Pillar 1 direct payments was assumed and costs for ancillary activities needed to support the delivery of environmental management in the agricultural and forestry sectors, such as advice and training, were not included. Importantly, in calculating these costs, allowances were made for the management of land to meet more than one environmental objective. The main overlaps identified related to biodiversity, resource protection and climate change. Where overlaps were identified, the cost estimate was based on the most expensive management option available (usually biodiversity focused).

The estimated total cost of meeting the UK's environmental targets in relation to **agricultural and forestry land management** over 16.2 million ha, was £1.986 billion per year, **averaging out at 122 £/ha**. This compares with current planned expenditure under agri-environment schemes in the UK of 742 million £/yr. Given the assumptions made and the lack of available detailed data for certain environmental issues and in certain regions, it is considered that these cost estimates are likely to be a significant underestimate of the funding needed in practice to deliver the UK's environmental targets through land management.

More recently, an IEEP study on "Costing the Environmental Needs Related to Rural Land Management[10]" led study (Hart et al, 2011) for DG Environment provided an estimate of the annual costs of delivering a full range of environmental objectives relating to biodiversity, soil protection, water resources, landscape protection and climate change on agricultural and forested land in 2020 (**Error! Reference source not found.**). The study followed a similar methodology to that used in the UK study by Cao et al (2009) and the soil restoration costs study outline above (Kuhlman et al, 2010) – see Box 2.1 for details.

Table 2-2 EU Environmental Targets used within the Rural Land Management Costs Study

Source: Hart et al, 2011

Environmental Issue	Target
Biodiversity	To halt the loss of biodiversity ... in the EU by 2020 (Decision of the European Council, 15 March 2010).
Water Quality	To enhance the status and prevent further deterioration of aquatic ecosystems and associated wetlands ... reduce water pollution and to achieve Good Ecological Status of all water bodies by 2015 (Water Framework Directive 2000/60/EC).
Water Quantity	To promote the sustainable use of water and to mitigate the effect of droughts (Water Framework Directive 2000/60/EC).
Soils	No formal EU target. Derived target: To protect and ensure the sustainable use of soil by preventing further soil degradation, including erosion, deterioration, contamination and desertification (from Thematic Strategy for Soil Protection COM (2006) 231 Final and 6EAP 1600/2002/EC).
Climate Change mitigation	To contribute to the reduction of EU greenhouse gas emissions by at least 20% below 1990 levels (EU Climate and Energy Package, 2008) by 2020.
Climate Change adaptation	No formal EU targets. Derived target: To increase the resilience of agricultural and forest habitats to adapt to climate change – in practice this is likely to require management also identified under the biodiversity objective
Landscape	No formal EU targets. Derived target: to protect and enhance the EU's traditional agricultural landscapes, to maintain landscape features and to conserve and appropriately restore areas of significant landscape value (from 6EAP 1600/2002/EC) .

[10] Hereafter referred to as the Rural Land Management Costs Study

The **costs of undertaking environmentally beneficial land management on agricultural and forested land in 2020 were estimated to be in the region of 34 billion €/yr**. Payments would rise towards this level over the period 2014-2020. To this figure were added the costs of three other elements considered necessary for the delivery of good environmental management. These are the provision of a basic per ha payment over 60% of the LFA, to contribute to the economic survival of environmentally sensitive farms, as well as an estimate for investments needed in physical infrastructure and advice and training. These added an additional 9 billion €/yr to provide an **overall estimate of 43 billion €/yr (+/- €8.5 billion)** as the approximate level of financial resources needed (from EU and national/regional sources) to deliver the EU's environmental objectives using incentive based measures. It is important to note that LFA and other existing CAP funding is not included in the cost estimates in this current study, as it is focusing on the estimation of additional costs required to meet Target 2.

A breakdown of the costs across land use types is provided in Table 2-3.

This shows that a large proportion of the costs associated with environmentally beneficial management on farmland and woodland were found to be associated with arable land (18 billion €/yr). These costs relate to approximately 40-50% of arable land which would be under one or more relevant farm management practices. A further 30% of the costs (10 billion €/yr) were associated with grassland management, covering approximately 70-80% of grassland in the EU-27. Of these figures, organic management (on arable and grassland) and in-field options on arable land, such as maintaining stubble over the winter, account for a large proportion of the overall costs (22% each).

Only 10% of the estimated costs (3.4 billion €/yr) related to the creation or management of woods and forests, but meeting this need would require a significant increase in the current levels of spending on forests.

Table 2-3 Cost estimates for different land uses by type of management from the Rural Land Management Costs Study

Source: Hart et al, 2011

Land Use	Cost estimate for environmental management needed (€ billion)		Proportion of total cost estimate (%)	Proportion of cost estimate for each land use (%)
Arable	**17.6**		**51.5**	
Organic Management		3.5	10.2	20.1
Integrated Farm Management		1.9	5.6	10.7
In field management options[1]		7.5	22.0	42.9
Non-productive options[2]		2.3	6.7	12.8
Conversion to grassland		1.7	5.0	9.8
Management of structural features[3]		0.7	2.0	3.8
Grassland	**10.1**		**29.5**	
Organic Management		3.9	11.4	38.1
Extensive management		5.3	15.5	52.0
Buffer strips/Grass Margins		0.2	0.6	3.0
Management of structural features[3]		0.7	2.0	6.9
Permanent Crops	**1.5**		**4.4**	
Organic Management		0.4	1.2	24.9
Integrated Farm Management		0.2	0.6	13.2
Extensive management		0.7	2.0	49.0
Buffer strips/Grass Margins		0.1	0.3	3.9
Management of structural features[3]		0.1	0.3	9.0
Rice	**0.1**		**0.3**	
Extensive Management		0.1	0.3	100
Wetland Habitats (Grazing Marsh, Peatland etc)	**1.5**		**4.4**	
Extensive management		1.3	3.8	90.8
Buffer Strips/Grass Margins		0.1	0.3	5.8
Management of structural features[3]		0.1	0.3	3.4
Forest and Woodland	**3.4**		**9.9**	
Management of existing woodlands		3.0	8.9	88.2
Creation of new woodland on agricultural land		0.4	1.0	11.8
Totals	**34.2**		**100**	

[1] In field options include reduced inputs, crop rotations, green cover, overwinter stubbles, spring cropping, reduced tillage, using rare or threatened crop species, and other soil protection measures.

[2] Non-productive options include fallowing part or whole fields, creating grass margins, beetle banks, buffer strips etc.

[3] Structural features include stone walls, ditches, banks, hedges, trees, terraces etc.

Box 2.1. Rural land use costs methodology

The methodology centred around three key steps as follows:

Step 1: Identification of the environmental pressures in rural areas, the form of land use on which the pressure is experienced and the area of the particular type of land use that is subject to the pressure. The main environmental pressures faced by rural land are well documented and were taken from an extensive literature review process. What was less clear-cut, however, was the area of different types of land that are subject to the different pressures. To derive these figures a review of the available literature was supplemented with expert judgment where data were lacking. Land use data were taken from the CORINE Land Use survey, Eurostat, and a report on the state of European forests for the Ministerial Conference on the Protection of Forests in Europe (MCPFE et al, 2007) to provide an accurate composite land use dataset allowing the identification of broad habitat types and farming and forestry land uses.

Step 2: Identification of the types of land management action thought necessary to respond to each of the pressures identified under Step 1 and an assessment of the area of rural land under pressure on which the management option is needed. There is a considerable body of evidence that provides information on the different types of land management activities (in agriculture and forestry) that are most appropriate to address each of the identified pressures. These were compiled based on two sources: a review of the relevant literature; and a review of a mixture of land management and investment measures adopted in a range of different Member States and included within their Rural Development Programmes (RDPs). A short list of the most significant management options for the provision of environmental benefits was then compiled, determined by taking those options that were able to deliver multiple benefits, those that were most frequently used by Member States in RDPs or those that were essential for the delivery of a specific environmental objective. These included only area based payments (excluding capital costs, advice and training) due to data accuracy and availability.

Assessing the area over which the desired form of management was needed proved much more problematic especially in a consistent manner across the EU-27. In particular it was recognized that environmental targets were often a reflection of what is possible using existing funding resources rather than an indication of the full extent of management required to meet the environmental priorities that formed the basis of the study. As a result the estimates of the area requiring management were derived using expert judgment.

In order to make the task manageable, a number of significant simplifying assumptions were made, for example the costs were estimated using an average cost per ha for individual management options, with the costs based on ‘income foregone plus additional costs’ as determined under the Rural Development Regulation. In reality, however, payments based strictly on this formula may not always be sufficient to stimulate sufficient adoption by farmers / land managers.

Step 3: Calculation of the costs of delivering specific voluntary management options over the estimated land areas in order to meet the suite of environmental priorities. Using the readily available Microsoft Excel a simple database was created in which the outputs of steps 1 and 2 were inputted. These included: the total area of each land use within the EU-27 (ha); the proportion of that land use under pressure (%); the proportion of the land under pressure requiring management (%); and the cost of the management actions used to meet those pressures (€). This logical and systematic approach enabled cross checking at each stage in the calculation with the opportunity to remove duplications where they occurred, for example where management options were identified as addressing multiple pressures, or where management options overlapped (ie situations where multiple management options could deliver the same outcome on the same area of land). The resulting calculation provided both the total area of land (also by type) that required management and the cost to provide such management.

Case studies were used to provide some indication of the costs associated with individual environmental issues. For example it was calculated that approximately **12 billion €/yr would be needed to halt soil organic matter decline in the EU27**, that **16-23 billion €/yr would be needed to maintain HNV farmland** (depending on the proportion of such farmland requiring support) and that approximately **1 billion €/yr is needed to halt biodiversity declines on arable land** (of which 854 million €/yr is estimated to be needed to halt declines of arable bird populations).

The estimates provided were comparable with the predicted current expenditure under agri-environment and other relevant measures operated through rural development policy. The study highlights, however, that it cannot be assumed that simply having sufficient budgetary resources available will lead to the environmental outcomes being achieved. Policy design and effective implementation are critical factors that will influence the cost of achieving the desired results. In many cases achieving changes in management practices also requires a change in attitude and approach to a farm's core business activities.

The final part of the study made a brief assessment of the implications of a range of possible future policy and economic conditions for the cost estimates provided. These included the implications of changes in commodity and input prices, changes in the regulatory baseline, in the architecture of CAP direct payments and in the design, targeting and implementation of environmental incentive schemes. The analysis showed that the interplay of these different factors is complex and that not all work in the same direction and concludes that further work in this area is needed, including an exploration of the trade-offs between regulation and incentive measures in the economic conditions of the coming decade.

2.9 Benefits of ecosystem restoration

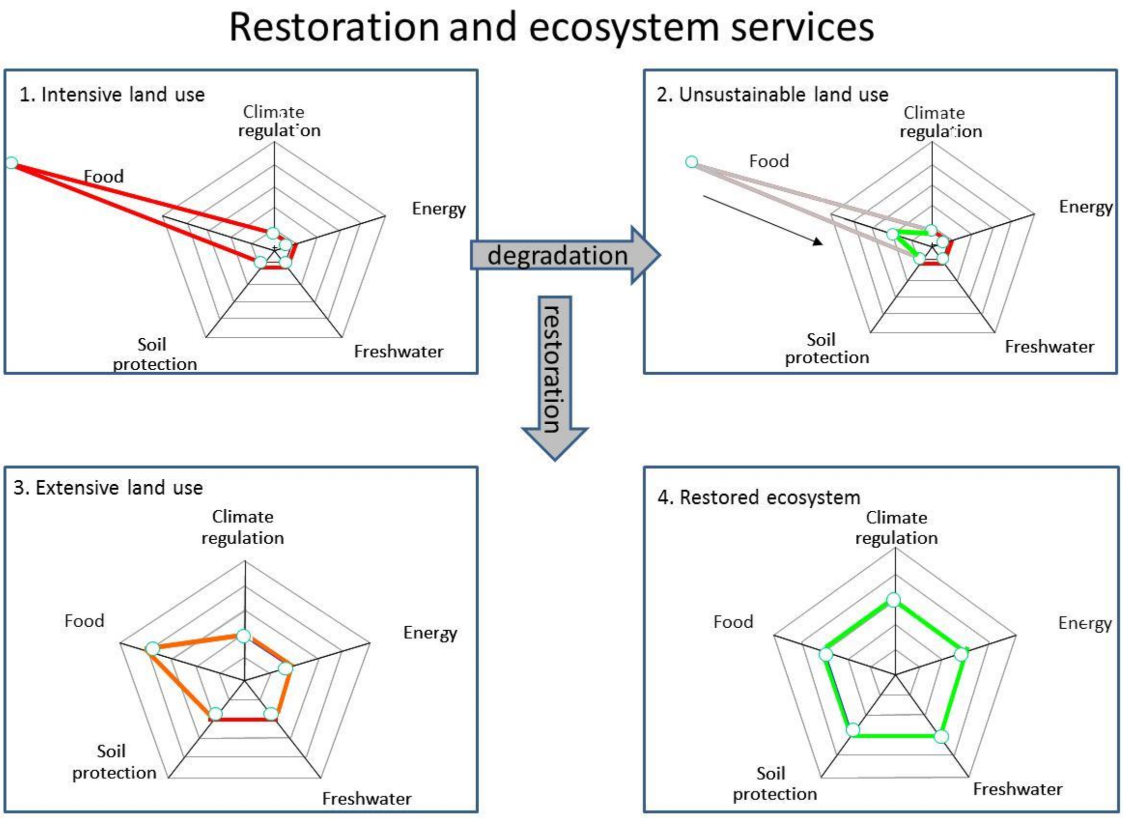
2.9.1 Rationale for focusing on ecosystem services

Natural ecosystems provide a range of services which are utilised by society (Millennium Ecosystem Assessment, 2005; TEEB, 2008; TEEB, 2010a; TEEB, 2010b; TEEB, 2011). Once the right priorities are set, the same area of land can frequently offer multiple benefits more efficiently than addressing needs individually, eg meeting biodiversity targets, reducing carbon emissions, provision of clean water, natural disaster reduction etc. As ecosystems, and especially their specific component habitats, vary in their importance for biodiversity conservation (eg in terms of their threat status, international importance, number of threatened species) and in their provision of ecosystem services, the value for money from restoration could be increased by targeting restoration to ecosystems and habitats that would provide the greatest benefits. In this respect it is important to note that CBD Target 15 for ecosystem restoration clearly gives a high level of importance to enhancing carbon sequestration and storage in ecosystems.

Restoration actions for nature conservation would undoubtedly contribute to the increased provision of many ecosystem services. However, a policy objective of focussing on increasing ecosystem service benefits would result in different restoration priorities. This would probably lead to more restoration of widespread and less natural ecosystems such as farmland and managed forests (eg to increase carbon storage and to protect soils and water resources). However, many service benefits (eg relating to water resources, pollination, soil erosion, protection from extreme events) are context and location specific. Thus what might be a priority in one country or local area in terms of ecosystem service provision might not be in another. Targeting of ecosystem restoration on the basis of generic ecosystem service values is therefore problematic (except for carbon, which has global-scale benefits) as a result of inadequate spatial information on ecosystem service values and beneficiaries. For such reasons the targeting of restoration according to ecosystem service may be best carried out at a national level.

Although ecosystem restoration can be expected to provide benefits to both biodiversity conservation and the delivery of ecosystem services (TEEB, 2011), there are, in many cases, inevitable trade-offs (Rodriguez et al, 2006). Figure 2-4 provides a simplified conceptual framework on the linkages between restoration and ecosystem benefits. An intensive form of land use (for example arable land) may deliver significant economic benefits on a single ecosystem service parameter (ie food production). It also prevents the ecosystem from delivering other services (such as climate regulation or water purification). In fact, the maximised benefit on a single service may be only temporary due to the unsustainability of intensive land use in an unbalanced ecosystem. Changing the land use to a more extensive form of land use, or restoring it to a natural state, increases the ecosystems capacity for a multitude of ecosystem services (eg flood protection combined with water purification and wood production), while assuring more sustainable delivery. Although the performance on a previously maximise service may decrease, the restoration increases the total value of portfolio of ecosystem services, resulting in net benefit from restoration.

Figure 2-4 Conceptual framework on restoration and ecosystem benefits



Source: modified from Ben ten Brink (MNP) presentation at the workshop: The Economics of the global Loss of Biological Diversity, 5-6 March 2008, Brussels, Belgium.

It is anticipated that the potential cost benefit ration of ecosystem restoration will be a key consideration in the establishment of a strategic framework for prioritising ecosystem restoration in the EU. However, although there is reasonable evidence of the benefits of maintaining existing ecosystem services, less information is available on the social and economic benefits of *the restoration* of ecosystem services. A number of studies have reviewed data on the costs and benefits of ecosystem restoration more broadly (Aronson et al, 2010; Bullock et al, 2011). These have shown that information on the benefits of restoration in the published scientific literature is relatively sparse and patchy. Nevertheless, it has been suggested that restoration benefits tend to outweigh costs, with cost-benefit ratios being in the range of 3 to 75 (Nellemann & Corcoran, 2010). But it must be borne in mind that many of the reviews have focused on ecosystems outside Europe which have no equivalents in Europe or which are significantly different in their ecological characteristics and processes. Therefore, it is difficult to obtain typical cost benefit ratios of restoration in the EU.

2.9.2 Ecological processes of restoration of ecosystem services

Currently, there are many uncertainties of the ecological processes underlying a successful restoration of ecosystem services is still relatively uncertain. Nevertheless, several key principles have emerged, identified by a recent review of the restoration of ecosystems and their services (Montoya et al, 2012). For instance, recent studies show that the historical sequence of colonisation can determine biological communities, which in turn affects ecosystem function (Fukami et al, 2010; Jiang et al, 2011; both cited in Montoya et al, 2012). Jiang et al (2011) found the assembly history of a community determines/affects its susceptibility to invasion by alien species. In addition, it has be suggested by Cadotte et al (2011) that functional diversity, described as the quantification of the similarities in phenotypes and ecologies of species (such as their environmental tolerances and how they impact ecosystem function), is a more important determinant of ecosystem function and service delivery than community diversity.

Other considerations of restoration of ecosystems and their services include the degree of modification from the original ecosystem state. In some cases, degradation is so severe that restoration to the original ecosystem state may not be possible or desirable. Indeed, TEEB (2011) distinguishes between three strategies to increase or enhance an ecosystem's capacity to provide services in the future: restoration, rehabilitation and reallocation. Rehabilitation is aimed at repairing some ecosystem processes at sites where one or more thresholds of irreversibility have been crossed, to recover the flow of ecosystem services but not to fully reproduce pre-disturbance condition and species composition. Reallocation means assigning irreversibly lost ecosystems a new function.

2.9.3 Evidence of the ecosystem services provided by pristine and restored ecosystems

Evidence of the improvement of (non-provisioning) ecosystem services as a consequence of restoration actions is still a relatively new discipline with a limited number of studies being able to show the impacts of actions over a period of time. Nevertheless, some studies have examined the impact over shorter time periods. In other cases, it will be necessary to consider the services obtained by pristine ecosystems.

For instance, peat soils of the boreal and cool temperate zones of northern Europe represent the single largest store of terrestrial carbon (EASAC, 2009). Restoration of wetlands, which have suffered particularly dramatic losses in the last century, has been widely attempted. A meta-analysis of 621 sites from around the world demonstrates that even a century after restoration efforts have taken place, biological structure (determined mostly by plant assemblages) and biogeochemical functioning (primarily the storage of carbon in wetland soils), remained on average 26% and 23% lower, respectively, than in reference sites (Moreno-Mateos et al, 2012). This could be due to either to slow recovery rates, or a move by post-disturbance systems to alternative states differing to reference conditions. Larger wetlands (greater than 100 ha) and those in temperate or tropical environments recovered more rapidly than smaller wetlands in cold climates.

Specific studies in the EU have been carried out. A study on a the rewetting actions and re-colonisation of highly degraded peat bog in Ireland (as a consequence of peat extraction) showed that restoration measures can result in sharp reductions in emissions of CO_2 on previously drained peatlands, and that appropriate management can restore the peatlands to a carbon accumulation status resulting in a net removal of CO_2 for the atmosphere (Wilson et al, 2012). From the point of rewetting, the bog mitigated an estimated 75 tonnes CO_2-eq/ha, resulting in an estimated value of 1,506 €/ha in avoided carbon losses plus an estimated an estimated 118 €/ha/yr in carbon sequestration. However, as the findings are based on monitoring data at one site over a 12 month period, a high degree of care is required in interpreting the data, and they cannot be necessarily applied to other sites. Factors likely to influence the carbon sequestration include trends in average rainfall, frequency/length of droughts. Plus it should be noted that restoration at Bellacorrick did not restore the peatland to the state that existed prior to peat extraction, and the vegetation communities are not typical of those of Atlantic peat. Nevertheless, the observed carbon sinks were greater than those typically associated with blanket bog (Koehler et al, 2011a; Koehler et al, 2011b).

It is important to note that vulnerable peatlands may become significant sources of GHG as a *consequence* of climate change; and therefore, the restoration of these ecosystems may result in a reduction in future emissions if restoration results in greater resilience. A study by Jones et al (2006) estimated that circa 40%% of Irish peatlands could be destroyed in the coming decades as a consequence of climate change, with partly degraded systems at greater risk. Measures may include the removal of colonising trees to prevent evapotranspiration and the maintenance of high water levels.

Restoration of peatlands can therefore bring economically valuable benefits in terms of mitigated carbon losses and carbon sequestration, which become more valuable as the price of carbon rises. Restoration also provides benefits for water quality and water regulation, as peatlands store large quantities of water (Förster and Schäfer, 2010). However, conditions will have to be monitored closely on a site-by-site basis to make sure that the desired change in services is in fact occurring.

Box 2.2 Estimates of potential benefits of peatland restoration

The Mecklenburg-Vorpommern (Germany) peatlands restoration strategy had restored an area of 29,764 ha by 2008, resulting in avoided emissions of about 300,000 tCO2-equivalents every year (Förster & Schäfer, 2010). Assuming a marginal cost of damage caused by carbon emissions of 70 €/tCO2, the value of peatlands restoration in the area amounts to €21.7 million every year, or 728 €/ha/yr (Mazza et al, 2012).

Natural England estimates that the restoration of key types of degraded peatlands in England could deliver emissions reductions of up to 2.4 million tonnes of CO2-equivalent each year, with 1.1 million tonnes of this reduction delivered by rewetting cultivated deep peatlands (Natural England, 2010). Comparing the costs of restoration of cultivated or agriculturally improved deep peat with carbon emission benefits, even the lowest shadow carbon value gives net economic benefits after 40 years of up to 19,000 £/ha.

A growing body of evidence is similarly developing of the benefits of restoration of freshwater habitats (Russi et al, 2013). For instance, wetland restoration was found to be the most cost-efficient measure to reduce nitrogen pollution in the Stockholm archipelago – providing a 20 SEK/kg nitrogen abatement cost, in contrast to the next cheapest measure of 25 SEK/kgN (Gren, 1995). The proposed restoration of riparian forest habitats (510 ha) along the Mesta River in Bulgaria would increase the sites' carbon sequestration capacity by 200 tonnes per year, to at least 1,036 t C per year or 3,797 t CO2 per year (EUC Coastal & Marine Union, 2010). In both case studies, carbon sequestration is only a co-benefit of habitat restoration, alongside other ecological and socio-economic benefits.

Acuña et al (2013), examined the impact of adding dead wood to restore stream channel complexity on the provision and value of ecosystem services in the Añarbe reservoir basin (N Iberian Peninsula), estimating net present values (NPVs) of fish provisioning, opportunities for recreation and tourism, water purification and erosion control. Results suggest that active restoration (i.e. actively adding logs into stream channels) greatly increases these ecosystem services and the additional benefits provided surpass the costs of restoration over realistic timeframes, which was not the case for passive restoration (i.e. allowing riparian trees to mature and fall into the stream). Initial results (after 2 years of monitoring) estimated a 10- to 100-fold increase in the monetary benefits provided by streams, constituting as much as €1.8 per metre of restored river length each year.

Box 2.3 An example of benefits of restored riverine systems

The Sigmaplan II is a long term strategy and list of projects to increase flood protection and promote nature restoration of the Scheldt Estuary in Belgium, through restoration of floodplains, estuarine nature areas and wetlands, creating over 5,000 ha of extra natural areas. The costs of the project up to 2030 are projected to be approximately €869 million, of which €247 million are attributable to nature restoration, and a further 100-200 €/ha/yr for ongoing nature management practices. It is expected to provide flood protection benefits of €740 million (over 2010 to 2100), additional ecosystem benefits (erosion protection, water retention, water purification, nutrient cycling, and carbon sequestration) worth €130 million and recreational values estimated at €22 million (Mazza et al, 2012). The flood protection benefits can be higher for specific cases, for example in areas at high flood risk and if initiatives are considered very effective to protect cities and industrialized areas.

Coastal saltmarshes host a distinctive biodiversity and provide important ecosystem services such as coastal defence, nutrient cycling, as well as acting as a nursery for fisheries. A case study in the Netherlands shows a benefit from dune water purification of 86 million €/yr in drinking water sales (which compares to 3.2 million €/yr for the site management) (Meulen et al, 2008). Rupp and Nicholls (2002) report that the benefits of salt marsh maintenance and restoration in the UK can be £4600 per metre coast thanks to reduced tide and wave impact resulting in avoided costs for maintaining sea defences. Turner et al. (2007) calculate the benefits of creating intertidal habitats as 421 £/ha/y, also taking into account the avoided costs for maintaining sea defences. Doody (2008) reports a benefit of £25 million from a case study in the Wash, comparing it to a cost of £2.61 million.

Nevertheless, a UK study by Mossman et al (2012) found that deliberately restored coastal saltmarshes displayed significantly different plant communities compared to reference sites linked to substantial functional differences. For instance, as the dominant species has a particular influence on productivity and nitrogen accumulation (Doherty et al, 2011), the plant composition is indicative of redox status of the soil (a low redox status is associated with higher greenhouse gas emissions) (Ding et al, 2010). Dominance by species such as Sea Purslane (*Atriplex portulacoides*) following restoration, as found on many of the sites examined, could inhibit colonization by rarer species that perform differing functions.[11] In addition, the restored sites would not satisfy the requirements of the EU Habitats Directive, indicating that meeting biodiversity objectives on restored saltmarshes remains a significant challenge.

[11] http://www.sfu.ca/~ianh/geog315/readings/zedler.pdf

Marine Protected Areas (MPAs), as part of a wider network of connected marine areas, can have positive effects on overexploited fish stocks, as well as support a range of regulating and cultural services. A first cut illustrative estimate for the benefits of increasing MPA coverage to 10% of the EU marine area is that this could deliver improvements valued at €2.5-3.8 billion per year in 7 services and €1 billion per year of off-site fisheries benefits (ten Brink et al, 2011).

The value of benefits delivered by the marine area currently protected by the network (equivalent to 4.7% of the EU's marine area) can be estimated to be approximately €1.4-1.5 billion per year. This would increase to €3.0-3.2 billion per year if 10% of the sea area were protected, and €6.0-6.5 billion per year for protection of 20% of the sea area. The higher figures apply to stronger protection measures.

This should be seen as an indicative value, to demonstrate the importance of this issue. To obtain more robust results would need an improved understanding of how protection will influence habitats, services and off-site fisheries; the level to which benefits will depend on details of protection; and network effects (Mazza et al, 2012).

3 APPROACH AND METHODOLOGY

3.1 Overall approach followed in this study

This study has two main objectives, firstly the calculation of the potential costs of achieving Target 2 and secondly the identification of potential ways in which measures that are needed to achieve the target could be efficiently financed. The methodology for the latter is a relatively straightforward literature and case study review and is described in chapters 8 and 9. In contrast the methods for estimating the costs of ecosystem maintenance and restoration are complex and are therefore described in detail below.

The main aim of the calculation of costs is to reliably establish the order of magnitude of **additional costs of achieving Target 2.** In this respect additional costs are those beyond expected expenditure to 2020 (such as on CAP funded agri-environment schemes) that will contribute to the maintenance and restoration of ecosystems. It also aims to reveal further data requirements and analysis that would be needed to increase the precision and accuracy of the costs estimates. These cost calculations have been carried out through the following steps:

1. Interpretation and clarification of the aims of Target 2 (see section 3.2 below).

2. Calculation of the costs of supporting actions, in particular those listed in the EU Biodiversity Strategy, as indicated in Box 1.2 (see chapter 4).

3. Assessment of the degree to which existing measures will address wide-scale generic pressures (eg air pollution and water pollution) and the potential additional cost of meeting Target 2 (see chapter 5).

4. Identification of the main ecosystem types in the EU (see section 3.3).

5. For **each ecosystem type** (see chapter 6):

    - Estimation of their area in the EU (largely drawing on CORINE Land Cover data, see section 3.4.1).

    - The identification of **key pressures** (ie the most important pressures that need to be addressed in order to maintain and restore the ecosystem) affecting each ecosystem type and the percentage of the ecosystem significantly impacted by each key pressure.

    - Estimation of overall ecosystem degradation levels, with respect to:
        - 2010 baseline levels (or as close to 2010 as data allow);
        - 2020 according to a reference scenario, which takes into account existing policies and measures (eg under the CAP, WFD, and National Emissions Ceilings Directive) as well as anticipated changes in drivers of land use change.

- Identification of **key measures** (ie the most effective measures that are typically used to address one or more of the key pressures) and their individual unit costs.

- Calculation of the total costs of maintaining the ecosystem and restoring 15% of its degraded area, according to two methods, where data allow, based on:
  a. **overall rates of degradation** and the costs of addressing pressures through ecosystem-specific **combined measures** (ie packages of measures) that aim to maintain and restore the ecosystem; and/or
  b. specific estimates of the extent of each **key pressure** on the ecosystem and the costs of specific **key measures** to address them.

6. The calculation of the costs of maintaining and restoring all ecosystems according to various potential scenarios for disaggregating the overall 15% EU wide restoration target (see chapter 7).

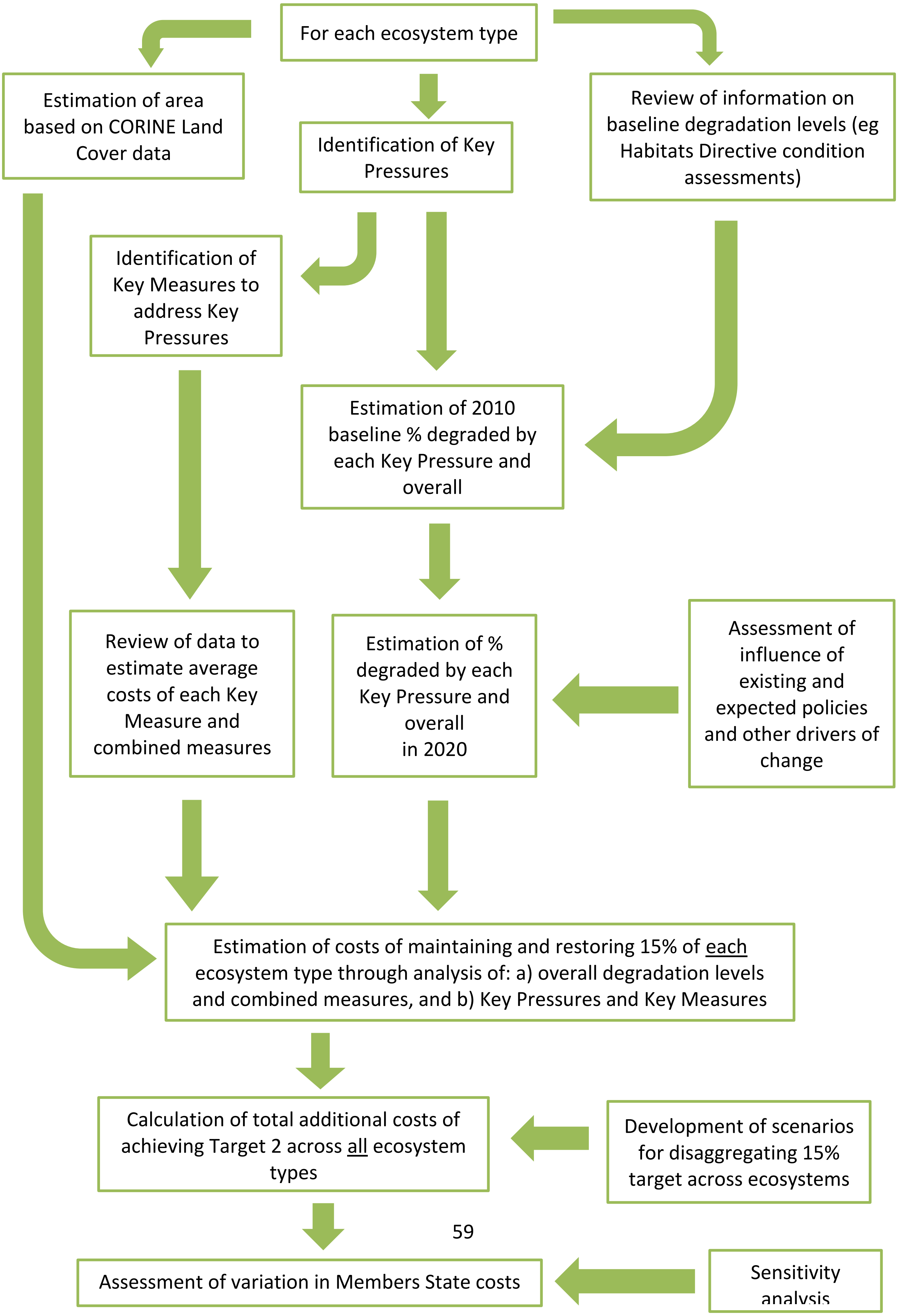
3.2 Interpretation and clarification of the components of Target 2

The interpretation of a number of aspects of Target 2, including the definitions of restoration and degradation, and the baseline against which degradation should be measured, have the potential to profoundly influence the results of this study. However, it is not within the scope of this study to make recommendations on these issues, as they will be considered as part of the development of the Restoration Prioritisation Framework (Action 6a), which is yet to be carried out by the Commission and Member States. Nevertheless, it is necessary to make some assumptions on the most likely future decisions on these issues in order to develop the cost estimates for this study. The interpretation of Target 2 is therefore further discussed below and key assumptions outlined where necessary.

3.2.1 What does the "15%" refer to?

It is firstly important to note that the target is ambiguous as to what the "15% of degraded ecosystems" refers to, and no official guidance is known to clarify its meaning. The simplest and probably most obvious interpretation is that it refers to 15% of all ecosystems that are degraded, which would probably then mean 15% of the EU as a whole because all ecosystems are degraded to some extent. However, such a target does not appear to be realistic unless a very narrow definition of ecosystem degradation is adopted. Therefore in this study **it is assumed that the target means 15% of those areas of each ecosystem type that are degraded**. For example, if 10% of an ecosystem is degraded then the target would result in restoration that is equivalent to 1.5% of the ecosystem's total area.

Figure 3-1 Overview of the calculation of costs of maintaining and restoring 15% of degraded ecosystems



3.2.2 What is restoration?

As discussed in section 1.1 the European Commission's definition of habitat restoration given in the Impact Assessment for the Biodiversity Strategy is extremely ambitious. In fact in most cases it will be impractical, and in many situations impossible, to achieve full restoration of ecosystems. In many cases full restoration would require measures to overcome the long-term impacts of some pressures, such as soil erosion, water pollution, acidification, nutrient enrichment and contamination with toxic substances. The full restoration of such areas would require very expensive and technically difficult actions, such as the removal of nutrient enriched or otherwise contaminated soils and sediments, and in some cases their replacement or augmentation with suitable soils. Furthermore, vegetation establishment takes time and some habitats will need to undergo natural succession processes to regain their original structures, ecological processes and composition. Consequently, the time taken to restore habitats to even a basic level is at a minimum normally decades and often hundreds of years (Morris et al, 2006; Perrow and Davy, 2002); whilst full restoration as defined in the Commission's Impact Assessment would take thousands of years if it is at all feasible.

In addition, restoration will be constrained by the absence of component species. Some might be able to recolonize given the right conditions, but some may no longer be in locations that can be a source of recolonisation. Translocations would therefore be necessary to restore the original complement of species, but would be prohibitively expensive for all but a few key species. In some cases restoration may be constrained by the global extinction of some species.

Lastly, it is reasonably certain that all these constraints on restoration will be exacerbated by climate change (Berry et al, 2007; Maltby, 2010; Moss et al, 2009). Although the exact impacts of climate change on biodiversity in Europe are difficult to predict it is clear that many habitats will change as a result of the differing responses of their component species to changing climate conditions and their indirect impacts. Thus some habitats may disappear completely whilst new habitats may arise with time. Consequently conservation objectives will need to be increasingly flexible and less based on maintaining particular species communities (eg as defined in the Habitats Directive) and more focused on providing space and favourable conditions for ecological processes and species (Hodgson et al, 2009; Lawson et al, 2012; Wilson and Piper, 2008). For such reasons, in all but a few exceptional circumstances, habitats cannot be feasibly fully restored, at least within reasonable time-scales and budgets.

There is also empirical evidence to support this view. In a review of 89 restoration projects across a range of ecosystem types globally (Rey Benayas et al, 2009), it was found that biodiversity measures for restored habitats were on average only 86% of the values for undamaged systems which, although much higher than the 51% for non-restored ecosystems, does not represent full rehabilitation. See section 2.9 for further discussion.

Where data allow, this study has therefore estimated Target 2 costs on the basis of the restoration of the key species, properties and processes of ecosystems and their functions. This should ideally take into account the concepts of functional diversity, as for example promoted by Cadotte et al (2011). This interpretation of the definition of restoration is also compatible with other definitions, perhaps most importantly with respect to Aichi Target 15. According to a guide on the target produced by the CBD (2011) "restoration is the process of actively managing the recovery of an ecosystem that has been degraded, damaged or destroyed as a means of sustaining ecosystem resilience and conserving biodiversity." This appears to be taken from the Society for Ecological Restoration (SER), a renowned international authority on restoration, which simply defines restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SERI, 2004).

However, the CBD and SER do not in fact define restoration, but describe the process in a rather open manner, and as a result the intended end point is uncertain. This is not explicitly acknowledged by the CBD, but it does refer to the SER primer on restoration that lists the following nine attributes that they suggest provide a basis for determining when restoration has been accomplished (although the full expression of all of these attributes is not essential to demonstrate restoration).

1. The restored ecosystem contains a characteristic assemblage of the species that occur in the reference ecosystem and that provide appropriate community structure.
2. The restored ecosystem consists of indigenous species to the greatest practicable extent. In restored cultural ecosystems, allowances can be made for exotic domesticated species and for non-invasive ruderal and segetal species[12] that presumably co-evolved with them.
3. All functional groups necessary for the continued development and/or stability of the restored ecosystem are represented or, if they are not, the missing groups have the potential to colonize by natural means.
4. The physical environment of the restored ecosystem is capable of sustaining reproducing populations of the species necessary for its continued stability or development along the desired trajectory.
5. The restored ecosystem apparently functions normally for its ecological stage of development, and signs of dysfunction are absent.
6. The restored ecosystem is suitably integrated into a larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges.
7. Potential threats to the health and integrity of the restored ecosystem from the surrounding landscape have been eliminated or reduced as much as possible.
8. The restored ecosystem is sufficiently resilient to endure the normal periodic stress events in the local environment that serve to maintain the integrity of the ecosystem.
9. The restored ecosystem is self-sustaining to the same degree as its reference ecosystem, and has the potential to persist indefinitely under existing environmental conditions. Nevertheless, aspects of its biodiversity, structure and functioning may change as part of normal ecosystem development, and may fluctuate in response to normal periodic stress and occasional disturbance events of greater consequence. As

[12] Ruderals are plants that colonize disturbed sites, whereas segetals typically grow intermixed with crop species

in any intact ecosystem, the species composition and other attributes of a restored ecosystem may evolve as environmental conditions change.

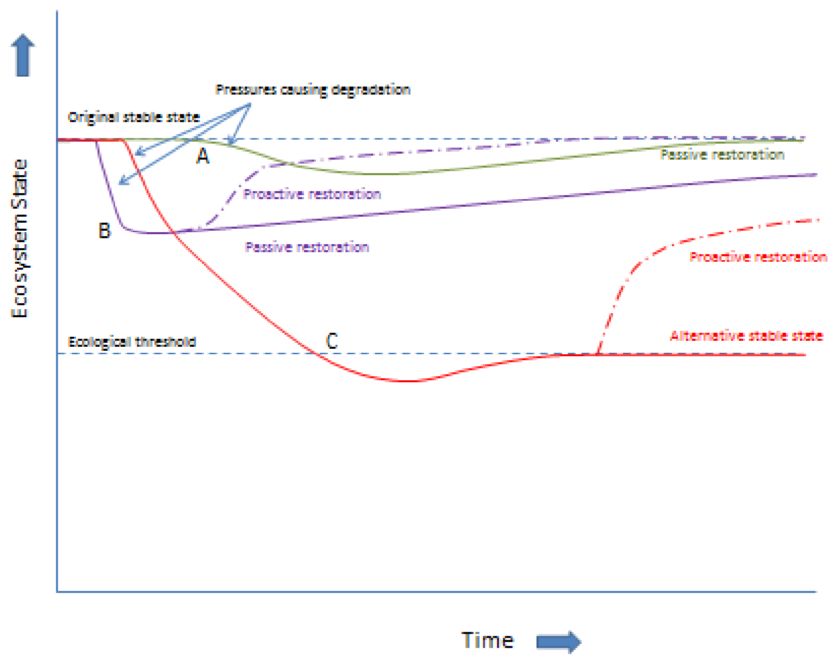
In practice the definition of restoration has had a limited impact on this study because available information on restoration costs (see 3.4.3 below) rarely indicates the degree of restoration that is aimed for or achieved. **It has therefore been necessary to assume that the observed costs of restoration that have been documented in this study provide a reasonable estimate of typical levels of restoration – although these are not defined**. This issue is further discussed in chapter 10.

3.2.3 What is a degraded ecosystem?

To estimate the likely costs of achieving Target 2, it is also necessary to understand what a degraded ecosystem is. However, unfortunately neither the EU Biodiversity Strategy nor the CBD provide a definition of degradation with respect to their targets. This makes it difficult to quantify the area over which restoration will need to be carried out for each ecosystem, and consequently the costs of achieving Target 2. The SER (2004) states that “degradation pertains to subtle or gradual changes that reduce ecological integrity and health.” This definition is therefore taken into account in this study, but as illustrated in Figure 3-2 degradation can be much more complex. In example A, degradation may result gradually as a result of slowly increasing pressures (eg nitrogen pollution or grazing levels) and if such changes are not too great in magnitude passive restoration may be possible by simply alleviating the pressures.

Figure 3-2 Illustrative examples of ecosystem responses to disturbances and restoration measures

Source: adapted from van Andel and Aronson, (2006)



However, as illustrated in example B, changes in pressures can be sudden and substantial. For example agricultural improvements, such as the drainage of a wet semi-natural pasture results in immediate profound changes in vegetation. Furthermore, such improvements also often trigger others. For example, the drainage of wet grassland allows the use of machinery and therefore ploughing and reseeding with more nutritious grass cultivars, which in turn makes the application of fertilisers and herbicides economically worthwhile. Thus a wet semi-natural pasture can suddenly become highly modified grassland with significantly enhanced agricultural productivity but with degraded biodiversity and associated ecosystem services. If such improvements are short-lived and not maintained then passive restoration may be possible, but this is likely to take a considerable amount of time without proactive restoration measures, such as blocking of drains and vegetation re-establishment measures.

A particularly important effect of substantial sustained pressures are that, as illustrated in example C, they may be sufficient to push an ecosystem past an ecological threshold, resulting in an alternative stable state (van Andel and Aronson, 2006), for example as a result of a build-up of nutrients in the soil and/or the loss of the former soil seedbank. In such cases restoration is only possible with proactive measures, that often require substantial and expensive changes to abiotic conditions (eg removal of highly fertile layers of soil) as well as biotic measures such as reseeding/replanting.

Defining degradation is therefore not straightforward, and requires careful analysis of the specific ecological requirements and conditions of each ecosystem type. However, such issues have been considered implicitly as part of the definition of Favourable Conservation Status for habitats of Community interest (see Box 3.1). It therefore seems appropriate to use this as a basis for defining degraded habitats in this study. This also has practical benefits because the Habitats Directive related monitoring data on habitat conservation status are the most comprehensive and standardised data that can be used to assess degradation levels. They also form the basis for the Biodiversity Baseline report (EEA, 2010a), which provides the most recent overall EU-wide assessment of the status of biodiversity. Although in some circumstances some habitats could conceivably have a Favourable Conservation Status whilst at the same time being degraded in terms of the provision of ecosystem services (eg carbon sequestration) this is unlikely to be a widespread situation.

We therefore assume that ecosystems that primarily comprise natural and semi-natural habitats of Community interest are degraded if their habitats have an unfavourable conservation status as defined in the EU Habitats Directive and related guidance.

Box 3.1: Article 17 reporting of the conservation status of habitats and species under the Habitats Directive

Favourable Conservation Status can be described, in simple terms, as a situation where a habitat type or species is prospering (in both quality and extent/population) and with good prospects to do so in future. Its more formal definition, under Article 1 of the Directive, take into account parameters including the 'extent of the area in which the habitat or species is found, the surface of the habitat area, its structure and functions (in case of habitat), the size of the population, its age structure, mortality and reproduction (of species)' (ETC/BD, 2006).

Under Article 17 of the Habitats Directive, Member States must report every six years on their progress in implementing the Directive, including on the conservation status of habitats and species of Community interest within their territories. To facilitate this a common assessment methodology was established in 2005, followed by supplementary guidance in 2006 (ETC/BD, 2006) to ensure a standardised reporting methodology. Nonetheless, differences in the way the data were collected and presented caused difficulties in presenting an overview at the EU level (ETC/BD, 2006).

Based on the reporting information, the Commission is required to produce a composite report including an overview of the conservation status of habitats and species of Community interest. The first ever reports were submitted in 2007 and 2008, covering the reporting period from 2001 to 2006, and included the conservation status of every habitat type (216 in total) and species (1182 in total) covered by the Directive for the EU-25 (Romania and Bulgaria were not required to report). The Commission published its first composite report in 2009 (ETC/BD, 2006).

For the purposes of reporting, Europe was divided into seven land and four marine 'bio-geographic regions', based upon similarities in climate, altitude and geology. The Member States' reports were used to assess conservation status across these regions by weighting each Member States' assessments according to the proportion of that species or habitat found within the national territories. These results were then aggregated to give a single, integrated assessment for each bio-geographical region. In total, 701 habitat assessments and 2,240 species assessments were made at bio-geographic level. More information is available on the website (http://biodiversity.eionet.europa.eu/article17) of the Member State assessments of conservation status (including maps and data sheets) and a detailed technical report.

The assessment of degradation of artificial ecosystems, such as arable and permanent crops, agriculturally improved grasslands and forest plantations dominated by non-native species is more problematical. The conservation status of such ecosystems are not assessed under the Habitats Directive or in any other standardised way that can be used as a measure of degradation levels. But, more fundamentally, the concept of Favourable Conservation Status cannot be readily applied to such ecosystems because they are according to the definition above always clearly going to be degraded. Although such ecosystems may still be important for biodiversity all have greatly reduced species diversity and many species that remain have impoverished populations. Furthermore, most are dominated by non-native species (or cultivars) and are often not self-sustaining (requiring for example regular pest control nutrient additions and replanting/sowing). The ecosystems are intentionally manipulated to greatly enhance certain ecosystem services (ie provisioning services such food, fuel and timber) which often results in the reduction of others (such as water storage and purification, nutrient cycling and carbon sequestration). In some cases the manipulation of the ecosystem may even result in unsustainable ecosystem service flows, as a result of, for example soil degradation and loss. Therefore, this study does not attempt to define or quantify overall degradation levels for **artificial ecosystems (ie cropland, agriculturally improved grassland and plantation forests).** Instead, as further described in section 3.4.2

maintenance and restoration costs are based on the proportion of each ecosystem that is considered to be at risk of, or affected by, specific key pressures (such as soil erosion).

3.2.4 What are appropriate reference ecosystems and baselines for setting restoration goals

A further issue to consider is what should be the reference ecosystem (van Andel & Aronson, 2006) that measures should be aiming towards restoring and maintaining. This is not straightforward in Europe, because virtually all ecosystems are highly modified as a result of human actions. Some might argue that reference ecosystems should be the historic natural ecosystems that would be expected to be present in a location in the absence of human influence, which would most typically be some form of climax forest. However, basing restoration on historically pristine ecosystems is not realistic as key natural components of such ecosystems are lacking in many situations (eg some large native herbivores and high level predators). Furthermore, widespread biophysical changes as a result of eutrophication, hydrological change and other factors now make it very difficult to restore many ecosystems, even to semi-natural systems.

There is also an argument that much of our biodiversity in Europe is any case now associated with semi-natural ecosystems (which also often provide a range of ecosystem services, especially those related to cultural values, recreation and health). Indeed many habitats of Community interest that the Habitats Directives aims to maintain and restore to Favourable Conservation Status are such semi-natural ecosystems. It would therefore be inappropriate to aim to restore all such semi-natural habitats to what are thought to be formerly more natural climax habitats. In some cases it might be appropriate for restoration goals to include increasing some former ecosystem types, but the extent to which this would be desirable at an EU scale is not known. Therefore, for the purposes of this study it is assumed that the reference ecosystems for degraded ecosystems that remain dominated by natural and semi-natural habitats of Community interest are their current type. Similarly where restoration aims to re-create[13] natural and semi-natural ecosystems that have been destroyed (eg by conversion to cropland) then the reference ecosystem should be that which formerly occurred. In other words, **in this study we assume that restoration of existing and former natural and semi-natural ecosystems is like-for-like**.

Appropriate EU restoration goals for heavily modified / artificial ecosystems, such as cropland and planation forests are not easy to identify from the wording of Target 2. However, because a substantial proportion of natural and semi-natural habitats of Community interest have been lost or fragmented by habitat conversion, it does seem reasonable to assume that a proportion of these should be re-created (ie reconverted). For example, agriculturally improved grasslands that were heathland could be returned to heathland. However, as indicated in Figure 3-2, ecosystem re-creation often requires active intervention measures to overcome ecological thresholds, which are often very costly, especially for highly modified ecosystems. Furthermore, since all areas of artificial ecosystem may be considered to be degraded then the target might be interpreted as requiring the re-creation of 15% of all converted natural or semi-natural ecosystems. The

[13] In this report we use the term re-creation to refer to restoration where the ecosystem has been destroyed in the restoration site

potential scale of restoration on this basis would be so large that it is clearly unrealistic, because of its direct costs, but also due to the large decline in the production of food and other products that would inevitably occur. This would result in policy conflicts and would inevitably be politically unacceptable under the current economic climate and at a time of increasing competition for land (Hart et al, 2013).

It therefore seems reasonable to identify an ecosystem conversion baseline against which ecosystem re-creation targets are set. However, it is not clear what that date should be, either with respect to the EU's Target 2 or the CBD restoration target which it relates to. Neither targets refer to a baseline date or cut-off date against which degradation should be measured. It therefore appears that the most logical and justifiable assumption for this study is to refer to the first EU headline target that aimed to halt the loss of biodiversity, which implies a baseline year of 2000. The specification for this study refers to the need to compare degradation levels with the EU Biodiversity Baseline report that was produced in 2010 (EEA, 2010a). But in fact, the data that the report was based on were mostly collected between 2001 and 2006 and therefore using a 2000 baseline would not be entirely inconsistent with the specification.

Although selecting 2000 as the baseline would result in relatively low levels of re-creation of natural and semi-natural ecosystems that have been converted to heavily modified / artificial ecosystems it would still result in considerable efforts to restore the condition of less degraded ecosystems, which we assume is the primary intention. **This study therefore assumes that the 15% restoration requirement under Target 2 includes the re-creation of 15% of ecosystems that have been ecologically degraded through conversion to another ecosystem type since 2000**, **as well as ecosystems that were considered to be degraded in 2010 according to the Biodiversity Baseline report** (see hypothetical example in Table 3-1).

Table 3-1 Hypothetical example of the calculation of the restoration requirements for a semi-natural ecosystem, based on overall area lost, overall degradation and degradation resulting from specific key pressures

	2000	2010	Expected in 2020[*1]	15% restoration target[*2]	Restoration required in 2020
Ecosystem area (ha)	10,000	8,000	7,000	>7,450	450

	2010	15% restoration target[*2]	Expected in 2020[*1]	% restoration required in 2020
Degradation				
Percentage degraded overall (ie in unfavourable conservation status)	60%	<51%	55%	4%
% degraded due to eutrophication	50%	<42.5%	40%	None
% degraded due to burning	10%	<8.5%	0%	None
% degraded due to under-grazing	6%	<5.9%	20%	14.1%

Notes: *1 According to reference scenario, ie taking into account expected trends in drivers and existing and forthcoming policies. *2 ie for area: 2000 baseline area minus 15% of area lost since baseline year (ie 15% x 3000 ha), or 2010 baseline 15% of area degraded in baseline year of 2010.

In addition to the restoration and re-creation of natural and semi-natural ecosystems described above, it is also assumed that restoration should also occur within heavily modified / artificial ecosystems to the extent that it is feasible. In other words, maintenance and restoration measures for such ecosystems would focus on ecological rehabilitation (van Andel & Aronson, 2006), or enhancement, rather than restoration *per se*. This should include reducing or halting degradation that imperils the long-term continuation of their primary goal of providing provisioning services. With such measures artificial ecosystems would remain degraded in ecological terms compared to other ecosystems, but degradation would be reduced as much as possible. In fact this concept is consistent with requirements for restoring heavily modified water bodies and artificial water bodies under the WFD. The Directive does not require the restoration of such water bodies to achieve Good Ecological Status as required for all other water bodies. Instead, in recognition of the high degree of abiotic modification of the water bodies, it only requires the achievement of Good Ecological Potential (see further discussion in section 6.10).

It is therefore assumed for this study that restoration goals within artificial ecosystems should include the maintenance and restoration of biodiversity and key ecosystem services to the extent that is feasible for the ecosystem in question. It should, however, be noted that a proper assessment of what is feasible in terms of rehabilitation / restoration for each highly modified / artificial ecosystem would require further research that is beyond the scope of this study. The assumed restoration goals for artificial ecosystems used in this study are therefore of a preliminary nature.

At the time of writing this report the Commission had outlined a Restoration Prioritisation Framework that recognises four levels of ecosystem degradation and potential restoration. This four-level framework has therefore been used to summarise in Table 3-2 the assumed restoration objectives described above in relation to each degradation level and ecosystem type. It should be noted that in order to calculate the cost of restoring each ecosystem type in chapter 6, we assume that 15% of the degraded component of each ecosystem type is restored (as indicated in the table). However, as described further in chapter 7, it may well be that the 15% target is disaggregated, in which case restoration might not be required for all degradation levels or ecosystem types within them.

Table 3-2 Summary of the restoration target assumptions used in this study in relation to ecosystem degradation levels and condition assessments

Degradation Level	Description	Ecosystems	Condition	Assumed management & restoration objectives
1	Satisfactory abiotic conditions, intact ecological processes and functions, comprises natural / semi-natural habitats and species with healthy populations	Natural and semi-natural ecosystems, including all habitats of Community interest	Favourable Conservation Status as defined under the Habitats Directive	Maintenance measures as required depending on the ecosystem and pressures on it
2	Satisfactory abiotic conditions and dominated by natural / semi-natural habitats, but disrupted ecological processes and functions, and declining diversity and key species		Unfavourable Conservation Status as defined under the Habitats Directive	15%* restoration to achieve Level 1 AND Re-creation of 15%* of the area of each ecosystem type converted to Levels 3 and 4 since 2000
3	Highly modified abiotic conditions, reduced ecological processes and functions, dominated by artificial habitats but retains some native species in stable populations	Heavily modified water bodies, arable farmland, permanent crops, improved grasslands and plantation forests	Good Ecological Potential achieved for Heavily Modified Water Bodies, equivalent standards for other ecosystem types need to be defined	Maintenance measures
4	Highly modified abiotic conditions, severely reduced ecological processes and functions, dominated by artificial habitats with few and/or declining populations of native species		Good Ecological Potential not achieved for Heavily Modified Water Bodies, or equivalent standards for other ecosystem types	15%* restoration (rehabilitation) to level 3 through enhancement of biotic conditions

Note: * or another disaggregated target agreed under the Restoration Prioritisation Framework.

3.3 Development of the ecosystem typology

This study aims to provide estimates of the costs of achieving Target 2 for each of the main types of ecosystem in the EU. An ecosystem typology has therefore been used in this study that is based on the CORINE Land Cover (CLC) classification as this is widely used in the EU, and most importantly has been used to develop standardised EU maps and quantitative estimates of land cover at a range of NUTs region scales. Thus estimates of per hectare costs of restoration can be scaled up using these data, to provide EU, national and regional cost estimates.

Our typology also takes into account current work under the Mapping and Assessment of Ecosystem Services (MAES) initiative, which is being undertaken to implement Action 5 of the Biodiversity Strategy. This is developing an ecosystem typology that builds on the Biodiversity Baseline Report classification (EEA, 2010a), which is based on CLC classes adjusted according to the European Nature Information System (EUNIS) habitat types where necessary (Maes et al, 2012). This will form the basic classification for ecosystem mapping at European and national levels.

We have therefore followed the current proposed MAES classification as far as is practical within this study, in order to facilitate future comparisons between the costs of ecosystem restoration and the results of MAES assessments. The proposed MAES classification is presented in Table 3-3 below, with an indication of the how this has been used and adapted slightly for this study. The main changes have been to sub-divide some large ecosystem types to reflect significant differences in restoration goals and actions. We have avoided making changes across groups. Therefore, the estimates from this current study can be combined for the sub-divided ecosystems to provide results that are directly compatible with the current MAES classification.

Inland dunes and limestone pavements are dealt with in this study under natural and semi-natural grasslands because their key management measures are similar (see section 6.4). Other sparsely vegetated ecosystems are not dealt with in this study because they are either very difficult to restore, and/or are relatively rare and will not significantly affect the cost estimates for achieving Target 2.

This study addresses ecosystem restoration needs and costs irrespective of whether the ecosystems are within urban or non-urban settings. Therefore there is no specific urban chapter within this report.

Table 3-3 The ecosystem typology used in this study and its relationship with the working typology developed by Maes et al (2012) for the Mapping and Assessment of Ecosystem Services study

MAES Level 1	MAES Level 2	Ecosystem type in this study
Terrestrial	Urban	Not covered separately
	Cropland	Arable ecosystems including temporary grasslands
		Permanent crops
	Grassland	Agriculturally improved permanent grasslands
		Natural and semi-natural grasslands
	Woodland and forest	Forest
	Heathland and shrub	Heathland and tundra
		Sclerophyllous vegetation
	Sparsely vegetated land	Not covered (although in practice inland dunes are partly covered under grasslands)
	Inland wetlands	Mires (bogs and fens)
		Inland marshes
	Coastal	Coastal ecosystem – beaches, dunes, saltmarshes, estuaries & lagoons
Freshwater	Rivers and lakes	Rivers and lakes
Marine	Benthic photic ecosystems	Specific costs for marine ecosystems are not calculated in this study
	Benthic non-photic ecosystems	
	Pelagic photic ecosystems	
	Pelagic non-photic ecosystems	

3.4 Estimation of key factors affecting ecosystem maintenance and restoration costs

3.4.1 Estimation of the area of each ecosystem type

The first step in the calculation of ecosystem-specific maintenance and restoration costs is the estimation of the area of each ecosystem type. This has primarily been carried out through an analysis of CORINE Land Cover (CLC) data because these are the only up-to-date source of standardised EU-wide land cover data available. Most of the used CLC data were published in 2006[14], and derived from satellite imagery taken between 2005 and 2007 (Büttner et al, 2012). Some more recent land cover data are available for some land cover types and/or areas, but this study only uses the CLC 2006 derived area data in order to provide a comparable data set, to avoid double counting land area, and to match other analyses that may be carried out using the CLC data in future (eg within the MAES working group).

It has been necessary to use expert judgement to assign land cover in the mixed CLC land cover classes to each ecosystem, and also to separate improved grasslands from semi-natural and natural grassland (see chapter 6 and Annex 1 for details on how this was done for each ecosystem).

These CLC data are able to provide reasonable estimates of the areas of most of the ecosystems types that are addressed in this study, but it is important to note that the data have some important drawbacks and therefore derived estimates of ecosystem extent need to be carefully interpreted. Ecosystem-specific CLC data considerations are discussed in the relevant sections of chapter 6, and where key alternative sources of data are available these are mentioned. But in summary the main problems concerning the use of CLC data in this study are:

- **Available CLC data from Greece are minimal,** with more than 99% of land cover unknown. Therefore the area of ecosystems that occur within Greece are likely to be underestimated, especially those that are particularly extensive in the country and concentrated in southern Europe, ie Sclerophyllous scrub**.** Furthermore, it is not possible to provide estimates in chapter 7 of the costs of meeting Target 2.

- **Fragmented ecosystems with small patches of habitat may be underestimated.** Although the CLC data are derived from satellite images with fairly high resolution (up to 100 m^2), the minimal mapping unit in the CLC dataset is fixed at 25 ha. This means that small patches of habitats, such as patches of grasslands within a predominantly wooded landscape, may be underrepresented in the CLC data.

- **Habitats of Community interest tend to fall within multiple CLC classes** Habitats of Community interest, as listed in Annex I of the Habitats Directive, are a focus of European biodiversity conservation policy, so restoration measures for these

[14] The UK data were published more recently, and the data for Greece are from 2000.

habitats are given particular attention in this study. However, there is no clear direct relationship between CLC classes and these habitat types. Instead, a single Annex I habitat can belong to several CLC classes (for example H1130 estuaries may be CLC class estuaries but also coastal wetlands, depending on the water depth; H5130 *Juniperus communis* formations on heaths or calcareous grasslands may be CLC class moors and heathland, grassland, transitional woodland scrub or even forest, depending on density of *Juniperus* vegetation).

- **CLC data splits closely linked ecosystems.**
  The area estimates of ecosystems are derived from CLC data instead of ecosystem functionality, therefore the areas of functionally coherent ecosystems are spread over multiple CLC classes. This means that restoration measures for one CLC class, may involve management actions in another CLC class. For example the restoration of estuaries may require intertidal habitat creation (which is covered by CLC class coastal wetlands), and the restoration of dune ecosystems may encompass restoration of dune forests, although these are defined as CLC class forests.

3.4.2 Estimation of baseline and 2020 reference scenario degradation levels and key pressures

A study commissioned by DG Environment (Slingenberg et al, 2009) identified the major direct drivers and resulting pressures causing the loss of biodiversity in Europe as:

- the conversion of natural habitats through land use changes, such as agricultural expansion and intensification, deforestation and infrastructure development;
- pollution from agricultural, urban and industrial wastes;
- unsustainable use of natural resources, for example, through forestry, fisheries and mining activities;
- the impact of climate change (eg fire, floods and droughts) and by making habitats unsuitable for some species; and
- invasion by alien species.

However, they and their impacts differ amongst the various regions and ecosystems within the EU. For example, forest ecosystems over the past 100 years have suffered mostly from over-exploitation and nitrogen and phosphorous pollution, whereas biodiversity loss in inland wetlands and coastal ecosystems has been mostly due to habitat change, pollution and invasive species (Slingenberg et al, 2009). For such reasons this study has quantified baseline and reference scenario degradation and pressure levels on an ecosystem-specific basis rather than according to an analysis of general pressures.

This study firstly attempts to estimate the baseline overall degradation levels and pressure-specific degradation levels for each ecosystem type. **For this study the baseline year is taken to be 2010**. These baseline assessments draw on available literature and quantitative monitoring data where available (as indicated in the ecosystem sections of chapter 6). Data sources that are as close to 2010 as possible are therefore used to ascertain these levels, but it should be noted that the most relevant data are those from the monitoring of habitats of Community interest under the Habitats Directive, which were collected between 2000 and 2006 and reported on by the Commission in 2007 and used in the EEA *Biodiversity*

Baseline Report in 2010 (EEA, 2010a). There is therefore in most situations a time discrepancy between the collected data and the reported condition of the ecosystem. However, as the magnitude of this discrepancy cannot be quantified we have used the best estimate of baseline conditions and we have not attempted to adjust them.

This study has then used the baseline estimates of overall and pressure-specific degradation levels as a basis for the development of **2020 reference scenarios**. These are primarily our judgements, which attempt to take into account the expected effects of land use drivers, policies, legislation (eg under the Birds and Habitats Directives and the WFD) and funding (such as CAP payments – see below) on the pressures and overall degradation values within each ecosystem type. The 2020 reference scenario estimates of overall degradation levels and pressure-specific degradation levels are then used in the calculations of ecosystem-specific maintenance and restoration costs as shown below.

As a result of the subjective nature of these assessments and uncertainties over trends in drivers and the future implementation and effectiveness of policies **the reference scenarios are highly uncertain, and should be only be regarded as indicative for the purposes of this study**. The variation in the assessments also has a large influence on the final cost estimates and therefore this study has attempted to estimate for each ecosystem type both the expected minimum and maximum levels of general degradation and pressure-specific degradation.

3.4.3 Estimation of the costs of maintenance and restoration measures

Combined versus measure-specific costs

It is well known that the costs of reaching conservation objectives can vary considerably in both space and time (Wätzold and Schwerdtner, 2005), for example due to variability in the costs of labour, materials, equipment and land. Therefore, to provide robust cost data that can be scaled up, this study has attempted to establish cost estimates for key maintenance and restoration measures that relate to specific ecosystem types and the alleviation or reversal of the impacts of specific key pressures on them (see step 5 in section 3.1). This detailed approach is used because, as shown by previous studies of restoration costs, overall combined ecosystem restoration costs vary considerably according to the actions that are undertaken and these vary according to context. Therefore the primary aim has been to compile comprehensive information for each main ecosystem type on the cost of specific key measures (eg maintaining grazing, blocking drains or removing alien species), rather than overall estimates of the costs of restoration projects.

However, it is also clear that much of the evidence on restoration costs only provides overall combined costs of restoration, rather than measure-specific costs. Such data on overall combined restoration costs have therefore also been collated and used to provide a second type of cost estimate; for comparison and as a fall-back option.

Assessment of additional costs

It is important to note that, because this study is attempting to calculate the **additional costs** of meeting Target 2, **payments that are already contributing to the maintenance and restoration of ecosystems are excluded from the cost estimates**. These include existing payments and those that are expected as a result of foreseen policy developments according to each ecosystem-specific 2020 reference scenario. It is therefore particularly important to note that the costs of existing CAP support measures, including Less Favoured Area (LFA) payments and direct payments under Pillar 1 (see section 8.2 for details) are not included in this study, although such payments may contribute to the continuation of some important farming management practices that help to maintain ecosystems and valuable habitat components.

The direct payment is granted to farmers irrespective of their production and is paid to the majority of farmers across the EU provide that they comply with conditions that aim to maintain a desired environmental reference level. The reference level is an extension of the polluter pays principle. It defines the dividing line between the level of environmental responsibility farmers are expected to assume at their own expense and those actions that may require some form of financial incentive (see OECD, 1998; Scheele, 1999; Cooper *et al*, 2009). The reference level is based on a mixture of environmental regulations that apply to all land managers whether or not they benefit from CAP payments, with considerable overlap between the two groups (Allen *et al,* 2012). At present the reference level for farmland management consists of cross-compliance[15], other national and regional regulations that apply at the farm level[16] and for recipients of agri-environment payments, requirements on the use of fertilisers and plant protection products defined by Member States. Any land management actions undertaken under Pillar 2 should only be remunerated where they go beyond the requirements of the reference level.

Therefore the cost calculations set out in this study rely on two important premises:

- the majority of farmers across the EU are receiving support under Pillar 1 through the single payment (which varies greatly amongst Member States, but currently averages 227 € per ha)[17], which includes conditional requirements[18] that in themselves can help to avoid ecosystem degradation; and
- that any incentive payments under Pillar 2 are made on top of the single payment and provide management beyond the associated reference level.

[15] Including both the conditions of Good Agricultural and Ecological Condition (GAEC) and Statutory Management Requirements (SMR). As defined for the 2007-13 RDPs in Art.5 and Annex II and Art.6 and Annex III of Council Regulation EC 73/2009, and as revised for 2014-2020 in the legislative proposals in Art.91 - 95 and Annex II of COM(2011) 628/3.

[16] Including any farm-level requirements for maintenance of permanent pasture under cross-compliance rules

[17] Either the single Payment Scheme for the EU-15 Member States or the Single Area Payment Scheme granted to farmers in the EU-12 Member States who joined the EU in 2004 or 2007.

[18] set out in Council Regulation EC No. 73/2009

Sources of cost data

Estimates of the costs of combined and individual key ecosystem maintenance and restoration measures were compiled and entered onto an Excel spreadsheet. Documented cost estimates included those from studies of the actual costs of individual projects, reviews of the cost of measures giving average, minimum and maximum values and fixed payments for measures (such as agri-environment payments). The estimates were mostly obtained using the following sources of information.

Literature review

Firstly the following literature sources were reviewed:

- previous relevant studies;
- examination of key journals, including Biological Conservation, Conservation Biology, Ecological Engineering, Journal of Applied Ecology, Journal of Restoration Ecology, Journal of Environmental Management, and the Journal of Forest Ecology and Management) and other key publications;
- online bibliographic database searches;
- the following online sources of specialist information on ecosystem restoration:
    - Society for Ecological Restoration (Europe Chapter) Knowledge Base on Ecological Restoration in Europe
    - Compiled intervention summaries at Conservationevidence.com
    - European Centre for River Restoration Knowledgebase.

Although considerable efforts have been made to obtain habitat restoration cost information the literature review revealed a number of significant data constraints on the analysis (Box 3.2). Most importantly, records of habitat restoration costs are rare in published peer-reviewed literature and approaches to costing are variable with some studies reporting only capital and labour costs whilst others report a single overall cost for the project (Bullock et al, 2011). Additionally, although there is a considerable body of knowledge on habitat restoration in Europe, much of the information remains in unpublished project reports and databases. For these reasons the additional data sources from agri-environment schemes and LIFE nature studies were of particular importance.

Box 3.2. Data constraints on the analysis

Scientific literature rarely contain cost estimates
Relatively few detailed data on specific ecosystem restoration costs are available in the mainstream scientific literature and approaches to costing are variable. Most studies on restoration focus on the degree of biodiversity success for different measures, but actual costs of the measures deployed are scarcely reported. In a review of over 20,000 restoration cases studies for the TEEB report, for example, only 96 were found to provide meaningful cost data (TEEB, 2011).

Reports on costs of measures are not easily accessible
It is believed that a considerable body of knowledge on costs of different restoration measures exists, for example with land managers organisations. However, these data are not readily accessible (often due to publication in a local language) or remain unpublished. Some Member States have established national averages of management costs for different habitats, for example within a frame of subsidy schemes for site management. In the Netherlands, for instance, costs of habitat management has been extensively debated between government and land management organisations, resulting in a report that provides widely agreed average costs for habitat management (Verheijen et al, 2009b). These data are split out in costs for different actions per habitat type and therefore provides a valuable resource for this study. However, comparable figures for other Member states were not found.

Costs vary between Member States
As habitat management often requires manual labour and machine handling, among the main components of cost for restoration actions are the deployment of staff/labour hours and the costs of machinery (fuel, capital costs, etc). As costs of labour and fuel vary widely within the EU (Eurostat reports up to a factor 25 difference in average labour costs between European (NUTS 1) regions), the cost of restoration measures for habitats are expected to show a significant regional variation as well. Therefore care must be taken when cost data for restoration are only available in one or a few member states.

Costs vary depending on the size of the area being maintained or restored
It is increasingly shown that economy-of-scale applies to site management and restoration costs. Armsworth (2011) shows that the size of a nature site area is the most important determinant of management costs for 78 small protected areas in the UK and that larger reserves offer costs savings over a set of small reserves of equal area. The costs per ha therefore have non-linear relationships with site area, such that protecting a 40 ha site would be expected to incur only double the costs involved in managing a 10 ha site. Indeed, data from the RSPB suggests that wetlands that are smaller than 100 ha cost up to 13 times more to manage per hectare than sites larger than 100 ha (Ausden, 2007).

As the level of habitat fragmentation differs between European regions, economies of scale might be an important consideration when scaling up case study data to a higher (European) figure on restoration costs.

The use of agri-environment data

Most of the estimates of maintenance and restoration costs used in this study come from agri-environment schemes, which have calculated the costs on the basis of income foregone and transaction costs according to Commission rules. These therefore provide a fairly reliable estimate of the costs of measures that are included in such schemes. However, the data also show that the costs vary considerably as a result of variations in the measures and their levels of ambition as well as differences in land management costs amongst Member States. It is therefore difficult to deduce from these datasets typical average costs of key measures.

The costs of measures needed to achieve environmental objectives on agricultural land relating to biodiversity and key ecosystem services in 2020 were assessed in detail in the

IEEP Rural Land Management Costs study (Hart et al, 2011), as outlined in Box 2.1. These objectives included the headline target of halting the loss of biodiversity. Although the study did not calculate the specific costs of meeting Target 2, in practice it covered biodiversity maintenance measures that would result in restoration. Furthermore, restoration measures were identified and included to achieve other ecosystem service objectives, and these would have also contributed to biodiversity restoration benefits. Therefore the estimated costs of achieving Target 2 on arable farmland, permanent crops and improved grasslands in this study primarily draws on the results of the Rural Land Management Costs study and associated detailed collated information on the costs of measures in Rural Development Programmes (RDPs) etc. However, more detailed and updated assessment of costs focusing on biodiversity needs are carried out for other ecosystems (including permanent crops, semi-natural grasslands, heathlands, mires and forests).

Agri-environments payments rates are calculated by managing authorities on the basis of 'income foregone plus additional costs' as determined under the Rural Development Regulation. In reality, however, payments based strictly on this formula may not always be sufficient to achieve the required management in practice, as has been shown by the low uptake of certain management options under agri-environment schemes in some Member States (Poláková et al, 2011). Low payment levels are most likely to constrain the uptake of measures on the most profitable agricultural land, especially if they involve substantial, large-scale and long-term change in farming practices. The implications of this on the interpretation of the results of this study are further in section 3.4.4 below.

<u>Life Nature project results</u>

To help overcome the lack of published data on habitat restoration costs, we supplemented the literature review with an investigation of data on the costs of restoration measures undertaken in LIFE Nature projects. According to a preliminary search of the LIFE database (using the filter keyword "restoration") almost 1,000 LIFE and the LIFE+ projects between 1992 and 2011 involved restoration, which corresponds to almost one third of all projects financed in the same period. The analysis of the LIFE nature projects was carried out in collaboration with The LIFE Nature Unit and Astrale (the external LIFE monitoring team).

A selection of completed LIFE Nature projects that involved restoration were analysed by Dr Ricardo Scalera (a member of this study team) for those ecosystems that were found to have relatively few available data on habitat restoration costs, namely forests in southern and south-eastern Europe and sclerophyllous scrub. For these habitat types, 30 LIFE projects were selected by further examination of the LIFE database and consultations with Astrale. Restoration costs were then extracted from the selected projects and analysed. Since detailed information on costs and relevant units were not available in all the associated reports, direct contacts with 19 project beneficiaries were also undertaken.

At the same time as this study was analysing LIFE nature project a similar study was underway by Astrale (Herremans et al, 2013) that was examining grasslands and river restoration cost. Some of the results of the study have been used in this analysis, but it was not available in time to be fully taken into account.

Expert review

The compiled data on maintenance and restoration costs have also been reviewed by a number of habitat conservation and restoration experts (see acknowledgements), who also supplied further information on relevant published and unpublished studies/reports as well as estimates of undocumented restoration actions that they have been involved in, or know of. These reviews focused on ecosystems that mainly had old, incomplete or unreliable estimates of the costs of key measures.

Recorded information on the costs and benefits of measures and estimation of typical costs

Identified relevant overall restoration and measure-specific cost data were summarised and added to an Excel spreadsheet, with the following information recorded where possible:

- Ecosystem type (see 3.3 above)
- CORINE Land Cover class
- Member State
- Habitat classification
- Habitat classes used in the study
- Habitats Directive Annex I type if known
- Key pressure(s) addressed by the measure
- Indication of whether it is a general combined measures or specific Key Meaasure
- Key measure type (if applicable)
- Specific actions undertaken
- Cost of measure in Euros (one-off estimate/average, minimum and maximum if provided)
- Unit (eg per ha, per km, per ha per year)
- Required frequency of use of the measure over 10 years
- Degree of restoration achieved for general restoration costs, if known:
    - Full restoration = as defined by the Commission, see section 1.1
    - Partial restoration = restoration of the key species, properties and processes of ecosystems and their functions (eg the establishment of a woodland on suitable or rehabilitated soils with appropriate hydrological conditions, with all the dominant plant species in all vegetation layers, and adequate representation of age classes, provision of standing and fallen dead wood, re-establishment of populations of key species of invertebrates and animals.
    - Basic restoration = restoration of the key components of the vegetation only (eg the planting of a woodland with the dominant tree species).
- Quantified benefits if provided in the source
- Date of cost information
- LIFE Project number (if applicable)
- Reference source details (if applicable)
- Indication of whether the estimate is from a project budget (ie a predicted cost), from the results of a project (ie, observed actual costs) or an agri-environment scheme fixed payment rate (ie based on calculations of average income foregone).
- Indication of whether the estimate is considered to be atypical, in which case it is excluded from the analysis

The collated data provide the basis for the calculation of the ecosystem-specific maintenance and restoration costs in chapter 6. It should be noted that, because the cost data are not complete or necessarily fully representative samples, then some discretion has been used in the calculation of unit cost of each measure, with for example atypical costs excluded from the calculations (and recorded as such in the spreadsheet). The resulting estimates of the costs of combined measures and key measures are not therefore simple arithmetic means, but attempt to reflect typical costs.

3.4.4 Limitations of this study

Costs that are not included

As discussed in chapter 2 some studies have attempted to partition costs according to whether they are direct realised costs (eg for design, planning, physical works, and administration etc) or opportunity costs, which may be compensated for, by for example agri-environment payments (see section 8.2). However, the separation of costs into such categories is complex and the quantification of opportunity costs is a very difficult issue, unless they have been calculated for compensation payments. **Uncompensated opportunity costs are therefore not taken into account in this study**.

This study focuses on the costs of practical ecosystem management actions because these are likely to account for the largest proportion of the total additional cost of achieving target 2. In addition the costs of specific new actions to support Target 2 listed in the Biodiversity Strategy are included in this study (see chapter 4). However, a variety of important supporting actions are associated with ecosystem maintenance and restoration activities, including:

- general ecological research and monitoring (which can help improve restoration methods and facilitate adaptive management);
- development of strategies, policies and legislation to support restoration measures, such as general measures to control alien invasive species (see section 2.4);
- administration regulations and funding;
- planning, development control and legal actions;
- advice and training, eg to support the implementation of agri-environment schemes.
- site wardening (eg to control visitors);
- regulatory enforcement; and
- communications, such as consultations, and awareness raising on biodiversity issues.

All these actions are currently being undertaken to some extent to meet existing EU and national biodiversity objectives. Therefore, although the achievement of Target 2 may require an increase in supporting actions such as those identified above, it is very difficult to ascertain the additional costs involved. This is primarily because many of these actions will also be contributing to other objectives, some of which might be closely related to the achievement of Target 2, such as other Biodiversity Strategy targets. Furthermore the existing costs of undertaking these actions for biodiversity are largely unknown. Although the costs of managing the Natura network have been estimated before (see section 2.2) this study is dealing with the whole of the EU environment and all ecosystems, including intensive farmland and urban areas. **Therefore this study does not attempt to estimate the**

implications of achieving Target 2 on the costs of general supporting actions for biodiversity conservation.

Additional important measures are often required to reduce ecosystem fragmentation, especially of lowland heathland habitats. This may require the restoration of habitat to increase the size of habitat patches and, where necessary to create habitat corridors and stepping stones to restore ecological connectivity amongst habitat patches. However, to avoid double counting of costs, **measures to specifically reduce habitat fragmentation are not further considered or costed in this study because they are likely to be addressed to some extent through the restoration of 15% of degraded areas of the ecosystem**, especially if strategically planned as part of ecological networks (Bennett and Mulongoy, 2006; Jongman and Pungetti, 2004) or through green infrastructure support measures (Mazza et al, 2012; Naumann et al, 2011).

Similarly **species-specific measures are not taken into account in this study**. These are measures that go beyond ecosystem maintenance and restoration requirements described in this study, such as measures to enhance or create specific types of habitat, predator or competitor control measures and species translocation costs. These are not included because the requirements for such actions and their costs are highly variable and therefore difficult to estimate reliably with respect to the achievement of Target 2. Nevertheless, it is very likely that the costs of realistic levels of species-specific actions will be a small fraction of the overall costs of restoring habitats to meet Target 2.

It should also be noted that **it is assumed that the purchase of significant areas of land is unnecessary to achieve Target 2 and therefore such costs are not included in this study.** This is because it is considered that the 15% target can be achieved through more cost-effective measures, such as regulations or funding incentives (eg agri-environment payments) that can be applied to private or state-owned land, whether this is managed for nature conservation purposes or not. Although land purchases may be required for some specific conservation objectives, such as to protect and/or manage particular sites, it is unlikely to be necessary to achieve Target 2.

The effects of demand and supply on costs

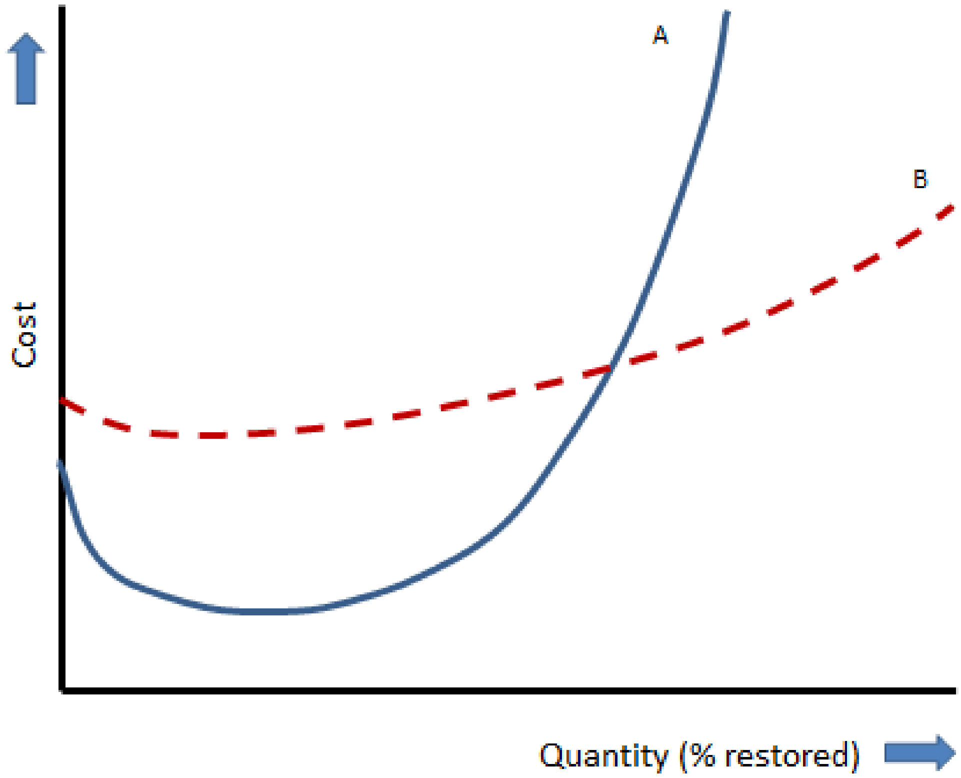
A significant constraint on the analysis undertaken in this study is that is not able to take into account the effects of potential variations in the demand for maintenance and restoration measures (which depends on degradation levels and restoration targets) and supply (that will depend on the availability of land that is suitable for restoration). In reality the marginal cost[19] of restoring each ecosystem is likely to vary according to curves such as those shown in Figure 3-3. Costs are expected to be high with low levels of demand (eg because knowledge is lacking and it is not cost-effective to build or use specialist equipment). But then marginal costs would be expected to decline as demand increases due to economies-of-scale. But with further increasing demand then it will become increasingly difficult to find suitable low-cost restoration areas and areas that require costly restoration measures will be increasingly required.

[19] Ie the cost of maintaining or restoring one more unit of each ecosystem

It is likely that the shape of cost curves will vary according to ecosystem types and levels of degradation, such that some areas of ecosystems (curve A) maybe relatively easily restored, such as through natural processes once pressures have been alleviated. But then costs may then rise steeply if more highly degraded areas that require more costly interventions need to be restored. This steep cost curve might be expected for many semi-natural ecosystems, and relate to restoration from level 2 to 1 (see Figure 3-3). In contrast, the costs of some ecosystems (dashed curve B) might be high initially (eg due to the high value of the land and the need for proactive actions), but restoration costs may change less if the same actions can be applied to the entire area of the ecosystem. This shallow cost curve might be typical of situations where highly modified ecosystems, such as arable farmland, are restored through enhancement measures (ie from level 4 to 3) rather than through major ecosystem changes.

Figure 3-3 Hypothetical marginal cost curves for the restoration of two example ecosystem types

See text for further explanation



However, although the likely shape of the cost curve is known, there is insufficient information available on the relationship between actual costs and supply and demand to quantify the relationship for any ecosystems. Consequently, it is important to note that this study is based on observed fixed agri-environment payment rates and other documented standard costs, and therefore **the cost estimates of meeting Target 2 do not vary in response to changes in demand and supply according to marginal cost curves**. In theory under market conditions (such as could be realised through auctioning) some restoration could be achieved with lower costs than suggested by the average fixed rates. Under such conditions the calculation of total cost, as carried out in this study, by multiplying the cost per unit area (under current 'equilibrium' output) by required area overstates total costs (which is the area under the cost curve). However, as discussed above, in practice agri-environment payments are not currently based on market rates and therefore any overestimation of the economic costs of restoration is largely theoretical. On the other hand, if disaggregation of the restoration target results in particularly high percentage restoration targets for some ecosystems (ie much higher than the overall 15% target), then costs of achieving the target for those ecosystems may be very high. Thus, the true costs of achieving the restoration for ecosystems with high targets under the scenarios presented in chapter 7 may be underestimated.

More importantly, **the costs of meeting Target 2 with regard to the maintenance of ecosystems is almost certainly significantly underestimated in this study**. This is because Target 2 requires maintenance over the entire land area, ie the far right of the cost curve. Thus very much higher payment rates than the current averages used in this study would certainly be required to even approach the target, let alone achieve it. Consequently it is unrealistic to assume that the full maintenance of all current areas of each ecosystem will be achieved.

The true cost of untargeted support measures

The cost estimates in this study are based on theoretical calculations that assume that the allocation of funds is in accordance with the theoretical need. For example, if it is calculated that 2% of an ecosystem is abandoned each year then the calculated annual cost of avoiding the abandonment (and thereby maintaining the ecosystem) is the average cost of the various measures that can avoid abandonment applied over 2% of the ecosystem. However, in reality, to achieve the aim of fully preventing abandonment it will not only be necessary to have high payments rates (for the reasons outlined above) but also to provide payments to many more people and hence a much larger area than is theoretically required. Maintenance cost estimates are expected to be the most affected by this issue because maintenance needs to be achieved over 100% of the area. Nevertheless, restoration costs may also be underestimated, especially in relation to Target 2 disaggregation scenarios that lead to high levels of restoration for a particular ecosystem (ie well beyond 15%).

Calculation of the actual payments that would probably be needed to achieve the desired results is a complex process as it will, for example, vary according to scheme design and demand and supply. Such calculations are obviously beyond the scope of this study and therefore **the estimate of costs should be treated as theoretical minimum costs that are likely to be higher in practice**.

The need for proactive measures to achieve restoration by 2020

A further consideration that should be taken into account in this study is that some restoration can be 'passive', when an ecosystem is able to regenerate through natural process alone (eg re-colonisation and succession) following the alleviation of pressures (eg damaging activities such as over-grazing) ; or 'proactive', which involves measures such as planting vegetation (see Figure 3-2). In many cases passive restoration is appropriate, but if restoration needs to be completed quickly, such as to meet the 2020 restoration target, then higher levels of proactive restoration may be required – with resulting higher cost implications. It is not possible to reliably ascertain from the information available the levels of proactive restoration that are required to meet restoration targets for each ecosystem in 2020. However, it is likely that **for the most degraded ecosystems a considerable amount of expensive proactive restoration would be required to restore them by 2020, and for these the costs of restoration are likely to be underestimated in this study**.

Revenues from ecosystem restoration

Restoration action can sometimes provide saleable products (eg timber, turf, top soil) that could be used to fully or partly offset their costs. However, restoration costs normally greatly outweigh income and estimation of the potential revenue that could be obtained or each ecosystem types is a complex task that is beyond the scope of this study. It is therefore assumed that in most situations revenue from maintenance and restoration actions is unlikely to be significant and therefore **all cost estimates in this study are gross costs**.

3.5 Calculation of the costs of maintaining and restoring each ecosystem

Once the required data on ecosystem extent, degradation levels, key pressures, and the costs of key measures have been collated and the baseline and reference scenarios defined, as described above, then the costs of maintaining and restoring **each ecosystem type** are calculated. The calculation is carried out two ways:

a. based on the estimated **overall level of ecosystem degradation** and average costs of **combined maintenance and restoration measures**; and

b. based on the **proportion of the ecosystem that is impacted by each key pressure** and the **costs of each key measures** that address each of the key pressures.

The two methods are used because it is considered that the use of overall degradation levels and generic combined costs may provide unreliable estimates for ecosystems, especially where there are considerable variations in the costs of component measures and their required use. The second method is therefore the preferred method, where sufficient data are available, because it is more detailed, transparent and realistic. Nevertheless, both methods are used where data allow, because the use of two calculation approaches provides a means of cross-checking the estimates.

The second method is more complex, and is therefore further described below using the moorland and tundra ecosystems cost calculation as an example (see section 6.6 for information on the derivation of the information used in the table). The cost estimation in Table 3-4 firstly establishes the additional costs of key measures that are expected to be needed to maintain the ecosystem and prevent degradation in the face of each on-going pressure in 2020 according to the reference scenario (ie taking into account expected changes in land use drivers, policy measures and funding etc). Importantly **maintenance**

costs are based on the expected annual rate of degradation resulting from each key pressure in 2020. The unit cost of each key measure is based on an assessment of the typical costs of the measure (from, for example, agri-environment payments rates or restoration studies) as discussed in each ecosystem section.

Each ecosystem account also considers the proportion of the area at risk from the pressure that the measure needs to be applied to. For example, grazing would normally be applied on 100% of an area affected by under-grazing. However, scrub growth may only be severe enough to warrant cutting (before grazing) over 10% of under-grazed areas.

The total cost of the use of each key measure is then calculated by multiplying the annual area of the ecosystem that is expected to be at risk in 2020 from each key pressure by the current cost of each key measure needed to address it, and then multiplying by the percentage of the area under pressure over which the key measure is needed. From the review of literature conducted for this study it is clear that the greatest area of uncertainty concerns the estimations of the extent of each ecosystem that is degraded by each pressure. For this reason minimum and maximum estimates of these pressures are used in the calculations.

Thus, in the example table below, the annual rate of agricultural abandonment of moorland and tundra areas is expected to be 0.1% to 1.2% in 2020 according to the reference scenario. This key pressure needs to be addressed through three key measures, the first being the maintenance of low intensity grazing. This particular key measure is always necessary and therefore needs to be applied to 100% of the area at risk of agricultural abandonment. Thus, taking the minimum pressure extent estimate of 20%, the cost of the measure is the area of the ecosystem, which is 8,949,182 ha x 0.1% x 116 € x 100%, which is €1,038,105. The other key measures are not required everywhere, but are estimated to be needed over 10% of the ecosystem area. As indicated in the 'instrument' column it is considered that the costs of these measures need to compensated for, and the principal means of doing this is through agri-environment measures (AEM).

The other key pressures that need to be addressed to maintain the ecosystem are over-grazing and inappropriate fires. However, it is considered that over-grazing can be fully dealt with through improved and enforced Good Agricultural and Environmental Condition (GAEC) regulations (as indicated in the 'instrument' column), and therefore there need be no compensated cost for this measure. GAEC regulations can also help address inappropriate burning, by requiring burning plans for intentional management burning. But it is considered that fire prevention and control measures will also be needed over the entire area at risk. The products of each line are then summed to provide the estimated total additional cost of maintaining the ecosystem in the 2020, which is about 2.2 million Euro.

The table then calculates the cost of **restoring areas that are expected to be degraded in 2020 through the cumulative impacts of each key pressure** according to the reference scenario. The calculation is carried out in a similar way to that described above for maintenance costs, but it is assumed for the ecosystem specific calculations in chapter 6 that restoration is only required on 15% of the degraded area. However, it is important to note that, as further discussed in chapter 7, the 15% restoration target is the overall target for the EU and this could be disaggregated such that it varies amongst ecosystems. The area to be restored also takes into account the area that is expected to be restored under the reference scenario (see discussion and examples in 3.2.4). Most restoration actions are only likely to be required once up to 2020, so the costs are normally one-off costs. Because the restoration actions may be carried out any time between this study's baseline year of 2010 and 2020 the costs are divided by 10 to provide an average cost up to 2020.

Finally the costs of re-creating ecosystems that have been converted to a different low biodiversity value ecosystem type are calculated. As noted in section 3.2.4 above, the year 2000 is taken as the conversion baseline. Costs are calculated by summing the percentage converted between the baseline year of 2000 and 2010 (using observed rates of conversion where available) and the area that is expected to be converted between 2010 and 2020 according to the reference scenario, and then multiplying 15% of the product by the per hectare re-creation costs for the ecosystem. It should be noted that re-creation costs do not include the costs of maintaining the area after re-creation, this is because such areas will be in the early stages of development and will therefore often not require typical longer-term maintenance management. As with the restoration costs, re-creation actions may be carried out any time between this study's baseline year of 2010 and 2020, and therefore the costs are divided by 10 to provide an average cost up to 2020.

Note that all 2020 costs are based on the most recently available price data and are not adjusted according to expected changes in measure costs or inflation rates.

Table 3-4: Projected additional costs (€) in 2020 of achieving Target 2 within heathland and tundra ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures and expected measures (ie the reference scenario) up to 2020. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4. Costs and totals are rounded to 3 significant figures. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Duplicate costs are removed where "below" or "above" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 8,949,182 ha.

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs (€) in 2020	
	Min %/y	Max %/y					Min	Max
	0.100%	1.20%	Low intensity grazing	116	100%	AEM	1,040,000	12,500,000
	0.100%	1.20%	Rotational burning	5	10%	AEM	4,470	53,700
	0.100%	1.20%	Mowing	248	10%	AEM	222,000	2,660,000
Agricultural abandonment and under-grazing	0.100%	1.20%	Scrub cutting	100	10%	AEM	89,500	1,070,000
Over-grazing	0.000%	0.000%	Grazing regulation			GAEC	-	-
	10.0%	10.0%	Burning management plans			GAEC	-	-
	10.0%	10.0%	Fire prevention & control	1	100%	AEM	895,000	895,000
Inappropriate burning and wild fires							-	-
MAINTENANCE TOTAL							***2,250,000***	***17,100,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% requiring restoration in 2020		Key measure	Sub measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs (€) over 2010-2020	
	Min	Max						Min	Max
Agricultural abandonment and under-grazing	1.45%	15.5%	Tree and invasive species removal		100	5%	AEM	64,900	694,000
Over-grazing	0.000%	0.000%	Vegetation re-establishment		75	2%	AEM	-	-
Inappropriate burning and wild fires	0.300%	1.50%	Vegetation re-establishment		75	5%	AEM	10,100	50,300
	0.300%	1.50%	Tree and invasive species removal		100	5%	AEM	13,400	67,100
Drainage and low water table levels	0.075%	0.750%	Hydrological restoration	Drain blocking	169	95%	AEM	108,000	1,080,000
	0.075%	0.750%		Turf stripping	71	5%	AEM	2,380	23,800
RESTORATION TOTAL								***199,000***	***1,910,000***

ADDITIONAL RE-CREATION NEEDS TO REVERSE CONVERSION SINCE 2000

Key Pressure	% area requiring re-creation in 2020		Key measure	Sub measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs (€) over 2010-2020	
	Min	Max						Min	Max
Afforestation	0.030%	0.105%	Re-creation	From forest	250	100%	AEM	67,100	235,000
Agricultural	0.000%	0.002%	Re-creation	From agriculture	562	100%	AEM	-	7,540
RE-CREATION TOTAL								***67,100***	***242,000***

4 COSTS OF SUPPORTING ACTIONS

In the calculation of the costs of habitat maintenance and restoration it is necessary to consider the possible need for additional strategic and supporting actions, such as research, consultations with landowners and stakeholders, management planning, the provision of advice and training for those involved in schemes and monitoring etc. Such actions are likely to provide broad benefits across ecosystem types and ecosystem services. Many will also be covered by existing environmental organisations and initiatives, but a significant increase in conservation and restoration activities may require additional institutional capacities and associated resources.

These supporting activities include a number of actions that were specifically identified by the European Commission in the *Biodiversity Strategy* as being necessary to achieve Target 2 (see Box 1.2). In this respect it is especially important to note that the establishment of a green infrastructure is an integral component of Target 2. Information on these actions and their costs from the available literature is summarised below. It is important to note that measures to deal with wide-spread pressures that are best dealt with at source (such as air pollution) are described in Chapter 4 and habitat-specific practical measures to restore the habitats themselves are dealt with in detail in Chapter 5. Therefore, to avoid potential double-counting of costs, such measures are not described in this chapter, which instead focusses on the costs of supporting actions (eg mapping, planning and the provision of guidance). These supporting costs are therefore likely to be very low compared to those of the wide-scale measures covered in Chapter 4 and the habitat-specific practical measures covered in Chapter 5. Nevertheless, their costs will be further examined and added to the cost estimates calculated in Task 4.

4.1 Action 5: Improve knowledge of ecosystems and their services in the EU

Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020.

4.1.1 Activities to be undertaken

This action aims to support the development of a prioritisation framework for habitat restoration, but also to enable environmental and economic impacts to be more fully taken into account in decision making. Activities include significant primary research and surveys, as the knowledge of ecosystem services is incomplete, especially in terms of the relationship between biodiversity and service provision, and the location of services and beneficiaries (EASAC, 2009; Maes et al, 2011).

Supporting activities include data handling, analysis (eg through GIS development) mapping and reporting. The incorporation of ecosystem service maintenance and restoration considerations into decision making is also a relatively new concept, which requires further development by policy researchers, and then on going implementation by environmental economists, accountants, and a variety of sectoral technical experts and managers (eg relating to coastal defences, flood management, water supplies, and climate).

4.1.2 Estimate of overall costs

It is difficult to assess the likely costs of this action as it is very broadly defined in the Biodiversity Strategy and some of the work is being carried out by Member States, which in some cases draws on previous completed or on-going studies. Nevertheless, although this is an ambitious and large-scale research contract its costs can be reliably assumed to be insignificant compared to the costs of more practical Target 2 related measures described below.

4.2 Action 6: Set priorities to restore and promote the use of green infrastructure

6a) By 2014, Member States, with the assistance of the Commission, will develop a strategic framework to set priorities for ecosystem restoration at sub-national, national and EU level.
6b) The Commission will develop a Green Infrastructure Strategy by 2012 to promote the deployment of green infrastructure in the EU in urban and rural areas, including through incentives to encourage up-front investments in green infrastructure projects and the maintenance of ecosystem services, for example through better targeted use of EU funding streams and Public Private Partnerships.

4.2.1 Activities to be undertaken

The costs of Action 6a are expected to be very limited as the action is primarily the development of tools by the Commission (with the assistance of external contractors) to help Member States make informed choices on priorities for ecosystem restoration. This is unlikely to amount to more than 0.5 million Euro, and is therefore relatively small compared to the other costs associated with meeting the target.

The costs of Action 6b are likely to be more significant as the recently published Green Infrastructure Strategy (European Commission, 2012) envisages that the EU policy action will result in an "enabling framework" providing a combination of policy signals and technical or scientific actions, implemented within the context of existing legislation, policy instruments and funding mechanisms. The Communication defines GI as "*a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, GI is present in rural and urban settings*." In this definition, green infrastructure is not only a nature conservation matter: all sectors influence green infrastructure and stand to gain from it. Therefore the conservation of protected areas is not sufficient for the maintenance of biodiversity and associated ecosystem services. Green infrastructure needs to be maintained and restored through proactive, strategic and coherent actions across all policies that influence the use of the land and sea.

The Biodiversity Strategy's emphasis on green infrastructure in Target 2 is a recognition that the spatial integration of measures in the landscape and the restoration of connectivity is essential. Restoration measures will therefore need to be spatially planned and implemented strategically to ensure their efficiency and coherence. While it is possible to identify key tools or policy instruments around which green infrastructure initiatives are structured, a wide set of implementing measures are almost always used, which may include the development of strategies and action plans, information gathering and mapping, spatial planning, regulation of land use (eg through SEA and EIAs), awareness raising and communication.

These requirements are recognised in the Green Infrastructure Strategy, which identifies certain key steps that need to be taken, including: promoting GI within the main policy areas, particularly in relation to funding; improving research and data, strengthening of the knowledge base and promoting innovation (including mapping and assessment); improving access to finance for GI projects; and supporting EU-level GI projects. Under the last heading a new initiative is proposed to develop an instrument – TEN-G – to promote large-scale cross-border GI programs with a pan-European vision (mirroring similar instruments for developing the trans-European networks in energy and transport).

The costs of practical actions that restore habitats to enhance green infrastructure are captured in the assessment of practical measures for each ecosystem in chapter 6. Therefore, the main focus here is on assessing green infrastructure costs that are associated with overarching measures that are relevant to the Green Infrastructure Strategy, and which are not included in the ecosystem cost estimates.

This analysis draws on a study by Mazza et al (2012) for DG Environment (2012), which examined the effectiveness and broad costs of different options for implementing a green infrastructure approach at the EU level. The study concluded that many actions are already underway, and many current EU and national policy instruments, exist that support practical activities that contribute to the development of a more effective EU green infrastructure. It is therefore not possible to provide a separate estimate of the current costs of implementing green infrastructure policy objectives. It also found that specific information on EU strategic level green infrastructure measures is largely missing and therefore most of the information on costs was obtained from project or national level sources.

Creation of a Gateway for European Green Infrastructure information

Two similar approaches in EU environmental policy to facilitate a broad information platform on specific topics already exist at the European level, namely

- WISE (The Water Information System for Europe) and
- BISE (Biodiversity Information System for Europe)

The on-off costs for such information systems can be generated from the respective tenders for the establishment of the systems. Both were contracted at a similar cost: for WISE the price range was fixed between € 230,000 and € 250,000 excluding VAT (including fees, travel and all other costs), while the budget of BISE is a maximum of € 200,000 excluding VAT (including fees, travel and all other costs).

It should be noted that these costs only include the development of the information platforms. Recurrent costs for updating, revising and managing the portals would also have to be taken into account when estimating the overall costs of such information systems. However, the overall amount is likely to be insignificant in comparison to other elements of green infrastructure implementation.

Awareness raising/ tailored guidance & technical assistance/ capacity building

These activities will need to occur at both the level of individual projects/strategies to ensure success on the ground as well as at the pan-/European level, to achieve integration into existing policies. At the European level, these actions will likely include:

- dissemination of evidence on the need and the benefits of green infrastructure projects;

- targeted stakeholder workshops on the overall concept of green infrastructure and its integration into specific sectors and relevant activities; and

- the provision of web-based information and data and the creation of networks of current and up-coming green infrastructure initiatives.

An indication of Commission costs for such activities comes from a recent contract on *Development of guidance document on management of farmland in Natura 2000 areas*[20] This was commissioned by DG Environment, and involved both targeted stakeholder workshops to key government officials and the production of official guidance documents, and had a maximum cost of €250,000 excluding VAT (including fees, travel and all other costs).

At the national and project level, the amounts spent on this measure vary quite considerably between initiatives, possibly reflecting the nature of the project or the design preferences of the project leaders. In the case of an initiative aiming at the protection and management of coastal habitats in Latvia, approximately 25 % of the entire budget, or €440,000, was spent on awareness raising. This was spent on public information boards, publication of leaflets, seminars and conferences, as well as target public awareness campaigns on specific habitat types and their importance. In the case of the Pumlumon farmland restoration project in Wales, €301,000 was set aside for the promotion of new tourism facilities and promoting the area as a tourist destination, approximately 10 % of the total budget (2012). In the Alpine-Carpathian Corridor initiative, communication or consultation constituted €440,000 or 9 % of the total budget (see Table 4-1). Therefore, the communication strategies at project level could equal exceed the cost of initiatives at the EU level to integrate green infrastructure into decision-making. This cost, taken together for initiatives, could be an important cost to consider alongside the practical restoration measures examined in chapter 6.

[20] http://ec.europa.eu/environment/funding/pdf/calls2010/specifications_en_10041.pdf

Table 4-1 Cost types of the Alpine Carpathian Corridor initiative

Cost type	€	% of total
Planning, surveys, preparatory studies	1,015,000	21%
Communication and consultation	440,460	9%
Project management and administration	205,000	4%
Spatial planning	67,500	1%
Land management and restoration works	130,850	3%
Creation of connectivity features	3,000,000	62%
Total	**4,858,810**	---

Source: Naumann et al (2011)

EU Green Infrastructure integration toolkit for spatial and regional planners

An example of this action is provided by the SITXELL project which started in 2001 as a project developed by the Barcelona Provincial Council as a territorial information system on the open areas in the province of Barcelona. It uses geographical information systems (GIS) with the objective of providing knowledge and raising awareness about the ecological and socio-economic value of natural areas, which would lead to a better consideration of natural values and ecosystem goods and benefits in integrated land planning. Between 2001 and 2010, the cost of SITXELL has been around €3,285,000, averaging to approximately 330,000 €/ha (Mazza et al, 2012).

At the project level, the total spend in four of the six in-depth cases which identified the cost of planning, surveys and preparatory studies ranged from €0.5m to €1.5m. There was no clear pattern with respect to what percentage of the total this action constitutes (see Table 4-2).

Table 4-2 Costs of planning, surveys, preparatory studies of green infrastructure initiatives

Cost type	Time period	Spend (€)	Total spend of initiative (€)	% of total
Alpine Carpathian Corridor	2002-2012	**1,015,000**	4,858,810	21%
Gallecs	2002-2004	**1,501,183**	1,501,183	100%
Transformation of the banks of the Rhone	2004-2008	**1,100,000**	42,886,000	3%
Tiengemeten	2008-2011	**530,548**	58,484,203	1%

Source: Naumann et al (2011)

Development of EU-wide 100x100 meter maps of green infrastructure

The closest example of this in the EU is the Czech Republic ecological network (Territorial System of Ecological Stability - TSES) initiative, initiated in June 1992, was designed to represent a network of ecologically significant segments of landscape, efficiently distributed on the basis of functional and spatial criteria, covering biotic, hydrological, soil and relief conditions. A map of existing and proposed bio-centres and ecological corridors was to be produced and to be used both for the development of TSES projects and for spatial planning in general, including with regard to land consolidations and land use change, processing of territorial planning documentation, forest management plans, water management and other measures affecting the conservation and restoration of the landscape. A rough estimate of the funding spent for the initiative as of January 2010 indicates that €10 million were spent for the development of a range of TSES plans, and around €4 million for the actual implementation, pre-dominantly covered by the national budget (Mazza et al, 2012).

4.2.2 Conclusions

Given the recent publication of the EU Green Infrastructure Strategy it has not been possible to assess its likely costs in this study. However, it was possible to examine some examples of the costs of a number of actions that appear to be relevant to the sort of measures proposed in the strategy. These suggest that overall the relative costs of implementing the Strategy at a European level are likely to be very small in comparison to the restoration measures carried out on the ground. However, it should be noted that the cost of awareness raising and capacity building, which is not always necessarily covered by the restoration measures, has been shown to be significant in numerous case studies.

4.3 Action 7: Ensure no net loss of biodiversity and ecosystem services

7a) In collaboration with the Member States, the Commission will develop a methodology for assessing the impact of EU funded projects, plans and programmes on biodiversity by 2014.
7b) The Commission will carry out further work with a view to proposing by 2015 an initiative to ensure there is no net loss of ecosystems and their services (eg through compensation or offsetting schemes).

4.3.1 Activities to be undertaken

Action 7a is already underway and being carried out through a separate DG Environment study on the development of a methodological framework for biodiversity proofing the EU budget, which is being led by IEEP. The study is currently finalising its final report and information from it will be used to assess the potential costs of implementing the proposed biodiversity proofing framework in the next Interim Report. However, it is excepted that additional costs will be relatively low as many of the key assessment tools are already widely used, such as Strategic Environmental Assessment (SEA) and EIA. Opportunity costs from more stringent biodiversity proofing could be significant, but these costs are not included in this study. Furthermore, such opportunity costs could be offset by economic benefits from avoided environmental damage and the more cost-effective use of funds (eg through ecosystem-based actions).

To support Action 7b, the European Commission is currently investigating the feasibility of developing a policy framework that would achieve no net loss of biodiversity. Although the scope of this policy and its governance is uncertain it is likely to involve some form of biodiversity offsetting (see Chapter 8). Biodiversity offsetting is a market-based conservation tool that aims to compensate for residual biodiversity loss (ie after appropriate mitigation measures to avoid and reduce impacts) incurred by development projects by maintaining an equivalent amount of biodiversity elsewhere that would otherwise be lost, or by restoring or enhancing biodiversity at an alternate location. Such offsets can be carried out on a project-by-project basis, or through habitat banks that may be set up in advance to offset anticipated impacts in a pooled approach.

The offsetting process typically involves buyers, sellers, and, in the case of habitat banking, third-party intermediaries. The buyers are developers requiring land for projects (eg housing), whereas the sellers are suppliers of the land to be used as an offset for the property to be developed. A range of third-party organisations, including local government, NGOs, insurers, brokers, traders and technical experts, may facilitate and regulate interactions between these two parties. A key component at the start of the process is normally the assessment of likely impacts by as part of an EIA. These are then compared with the potential benefits of possible offsetting options to ensure that they are ecologically equivalent (or better) and likely to result in no net loss of biodiversity, or a positive gain.

An important principle of offsets is that they should have long-term additionally (ie result in a benefit that would not otherwise have occurred). Therefore, although offsetting can in theory be carried out by protecting habitats and species that are at risk of future losses a more robust and defendable approach is to restore habitats and species that are unlikely to benefit from other anticipated conservation actions (eg management of Natura 2000 sites).

4.3.2 Estimation of overall costs

It is very difficult to estimate the costs of implementing a no net loss policy in the EU as its scope is currently unclear and it is uncertain how it will be implemented. Nevertheless, as it seems likely that a key instrument will be biodiversity offsetting the costs will probably fall on the project proponent. Thus public funding support would not be directly required to implement this action, other than to support the institutional capacity required for regulating and advising on offsets. The costs of such institutional support would depend greatly on whether offsetting is to be mandatory, its sectoral scope (eg whether it covers agricultural and forestry activities) and the policy and legislative framework that is promoted under the initiative.

The costs of the no net loss policy cannot therefore be estimated at this stage. However, it can be assumed that the policy would need to ensure that offsetting measures are efficient and proportionate and have reasonably low transactions costs. Consequently, the costs of Action 7 are unlikely to be significant compared to the other costs of achieving Target 2 considered in this study.

5 COSTS OF REDUCING WIDE-SCALE GENERIC PRESSURES AT SOURCE

5.1 The 15% target and the Water Framework Directive objectives

Before addressing the issues of point and diffuse pollution of surface and ground waters specifically, it is important to consider how Target 2 for ecosystem maintenance and restoration in 2020 relates to the legal objectives for waters that are currently in place and, therefore, whether Target 2 would be met by such objectives and whether additional costs may be required.

The 2000 Water Framework Directive[21] (WFD) requires that water bodies achieve 'good status' ('Good Ecological Status' for surface waters and 'Good Status' for groundwaters). The definition of status is derived from a combination of biological, chemical and hydromorphological criteria. Good Ecological Status requires water bodies to depart from 'reference' conditions to only a minor extent. Such reference conditions are effectively pristine conditions and the WFD requires the biological character to be determined according to a range of types of biota from phytoplankton and zooplankton to macrophytes, invertebrates and fish. Where water bodies have been heavily modified or are artificial (eg canals) the objective of the WFD is modified to that of 'Good Ecological Potential'. With regard to Target 2 this has implications for considering the maintenance and restoration of ecosystems either under changed conditions or in artificial circumstances.

The WFD also requires that Member States do not allow a deterioration in the status of water bodies. This is important in relation to the maintenance target within Target 2. On the surface, ensuring the maintenance of a given water body status (whether, high, good, poor, bad, etc.) would seem consistent with the Target 2 maintenance objective. However, water status is a 'banded' classification and, in theory, some deterioration could occur without a change in status. Having said this, there is no mechanism to report such a change in condition that does not change status at EU level and no analysis on the issue has been undertaken. Thus at this stage the stand still principle of the WFD is best viewed as consistent with the Target 2 objective.

The WFD is focused on 'water bodies' as water management units which are required to meet GES, which may be lakes, stretches of rivers, coastal waters, etc. Water bodies are not standardised across the EU, such as in relation to size. Member States are required by the WFD to restore water bodies to GES by the end of 2015. However, the Directive allows for exemptions, which effectively can result in a delay to the end of the 2nd river basin planning period (2021) or third river basin planning period (2027).

In relation to the biodiversity restoration target, two issues arise:

1. What is the relationship between achieving Good Ecological Status and whether biodiversity has been 'maintained' and 'restored'?
2. What degree of change in water status is expected by 2020, and would this meet Target 2 and/or would additional costs be required?

[21] Directive 2000/60/EC establishing a framework for Community action in the field of water policy. OJ L327

The first question effectively considers whether a water body maintained or restored to Good Ecological Status would be 'maintained' or 'restored' in terms of the biodiversity target. This is not a simple question, particularly as the biodiversity target is not elaborated in detail (as discussed in section 3.2). However, given the fact that the detailed ecosystem criteria within the WFD require only minor deviation from relatively pristine conditions, it seems reasonable to assume that a water body in Good Ecological Status is maintained or restored in accordance with the expectations under Target 2. For a water body to be at Good Ecological Status, significant pollution needs to be controlled, environmental flows ensured, etc.

The second issue is to determine the predicted changes in the status of water bodies. A study of River Basin Management Plans (RBMPs) submitted by Member States has been concluded and data from this study are included in a November 2012 Commission Communication on the implementation of the WFD (COM(2012)670). This contains the following data on the number of water bodies that have been reported on and the number and percentage that are expected to change from below Good Ecological Status or Good Ecological Potential to Good Ecological Status or Potential by the end of 2015.

	No of MS	**% water bodies Good Ecological Status or Potential 2009**	**% water bodies in Good Ecological Status or Potential 2015**	**Progress 2009-2015 in %**
Ecological status of surface waters	21[22]	43	53	10

Firstly, it must be stressed that these data only cover 21 Member States. 23 Member States have adopted and reported all their Plans. Four Member States (BE, EL, ES, PT) have either not adopted RBMPs or only adopted and reported some plans (eg only Flanders in Belgium and Catalonia in Spain have reported their plans). In total, the Commission has received 121 RBMPs for national parts of River Basin Districts (out of an expected 174[23]). Therefore, it might be assumed that expected progress for the EU as a whole could be less than that reported here. Overall, currently less than half of surface waters are at Good Ecological Status or Good Ecological Potential and a major improvement is required by law for 2015/2027.

A 10% improvement in Good Ecological Status or Good Ecological Potential is expected by 2015. However, this percentage is based on the overall number of water bodies. The 15% restoration target is based on restoration of those habitats which are degraded (ie not including those water bodies at Good Ecological Status or Good Ecological Potential in 2009). Taking this into account the percentage improvement for water bodies **NOT** at Good Ecological Status or Good Ecological Potential in 2009 is 17.5% for 2015 for surface waters.

[22] Ecological status: countries that have not reported RBMPs or not reported exemptions (DK) or high unknown status (IT) are not included.

[23] Norway has adopted 9 pilot RBMP. Norway is implementing the Water Framework Directive as part of the European Economic Area Agreement, with specific timetable agreed therein.

Furthermore, given the pressure on Member States to ensure compliance by 2027, it should be expected that further progress will be made in the 2nd round of RBMPs and further water bodies will be at Good Ecological Status or Good Ecological Potential by the end of 2012.

In conclusion, even with the missing RBMPs from a few Member States, it is expected that more than 15% of water bodies not currently at good status will be restored to good status by 2020.

The WFD requirement to not allow for a deterioration in water body status seems to be consistent with the Target 2 objective to ensure maintenance of ecosystems.

Therefore, achieving the Target 2 restoration target for water bodies and water courses will not result in any additional costs over and above that already required by the Water Framework Directive adopted in 2000.

The following sections explore the pressures on water bodies in more detail – considering issues of point and diffuse pollution, controls on these sources, costs of selected measures, etc. It must be emphasised that while specific controls, for example the Urban Waste Water Treatment Directive or Nitrates Directive, aim to tackle specific sources, the overarching integrating instrument is now the WFD. The UWWTD effectively sets minimum requirements – if achieving Good Ecological Status requires further action (eg on small sources), then additional action is needed. Thus the controls, measures and costs are illustrative and do not reflect any additional action needed to meet Target 2.

5.2 Eutrophication of water bodies from point-source pollution

5.2.1 Description of the pressure and proportion of habitats affected

Eutrophication of surface waters arises from the introduction of excess nitrogen and phosphorus from a variety of sources, including point sources (eg sewage work discharges) and diffuse pollution from airborne deposition and nutrient-rich run-off from farmland (see section below). Phosphorous is normally the limiting nutrient in freshwater ecosystem (and therefore the cause of eutrophication) whilst nitrogen is generally the limiting nutrient in marine water. Eutrophication radically changes ecosystem function. The excess supply of nutrients causes increased growth of key primary producers, such as algae, resulting in changes to zooplankton, invertebrate and vertebrate components of ecosystems. As primary producers die the resulting oxygen demand can de-oxygenate waters, so causing death of animals dependent on higher oxygen levels. A wide variety of habitats are at risk, most notably oligotrophic (low or very low nutrient) and mesotrophic (moderately low nutrient) waters, but added nutrients can cause changes in most freshwater and marine aquatic systems.

While industry can contribute to point sources of nutrients, by far the largest sources are waste water treatment plants (sewage) (Figure 5-1 and Figure 5-2).

Figure 5-1 Annual phosphorus discharges by source (EEA, 2005)

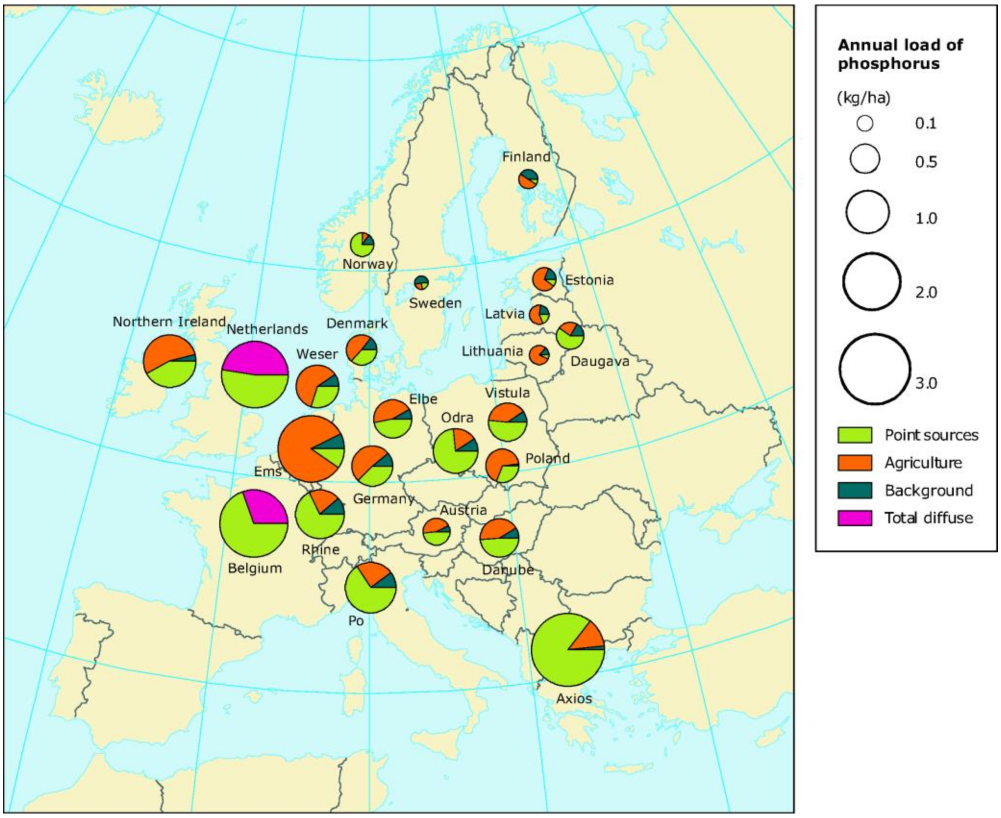


Figure 5-2 Annual nitrogen discharges by source (EEA, 2005)

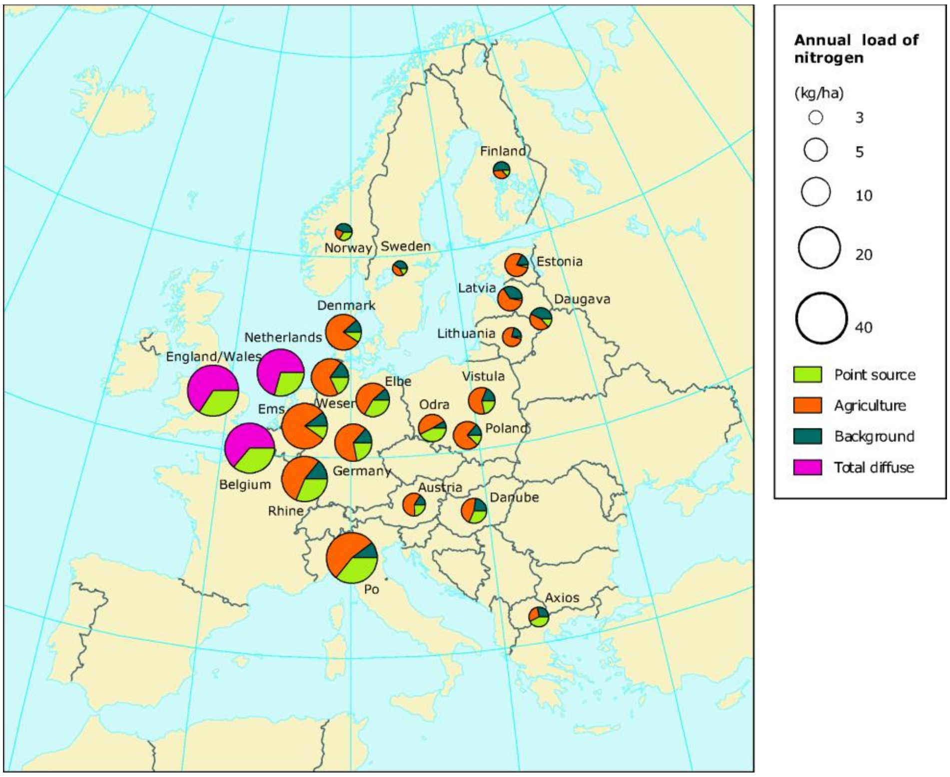


Figure 5-3 Oxidised nitrogen concentrations in transitional, coastal and marine waters in 2008 (EEA, 2011)

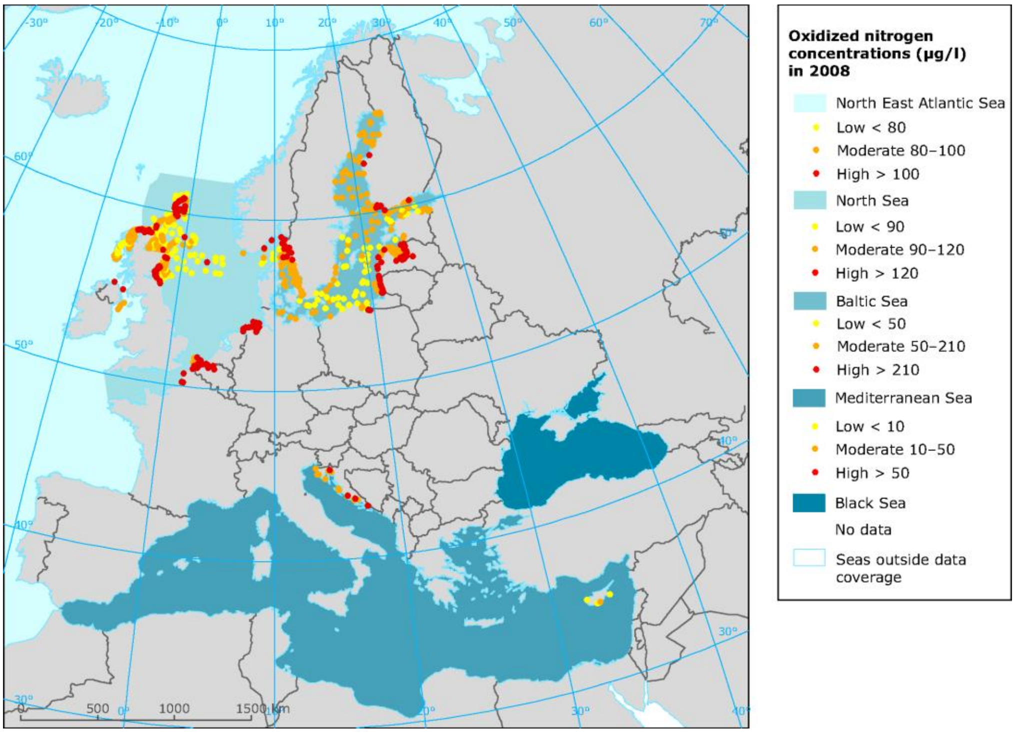
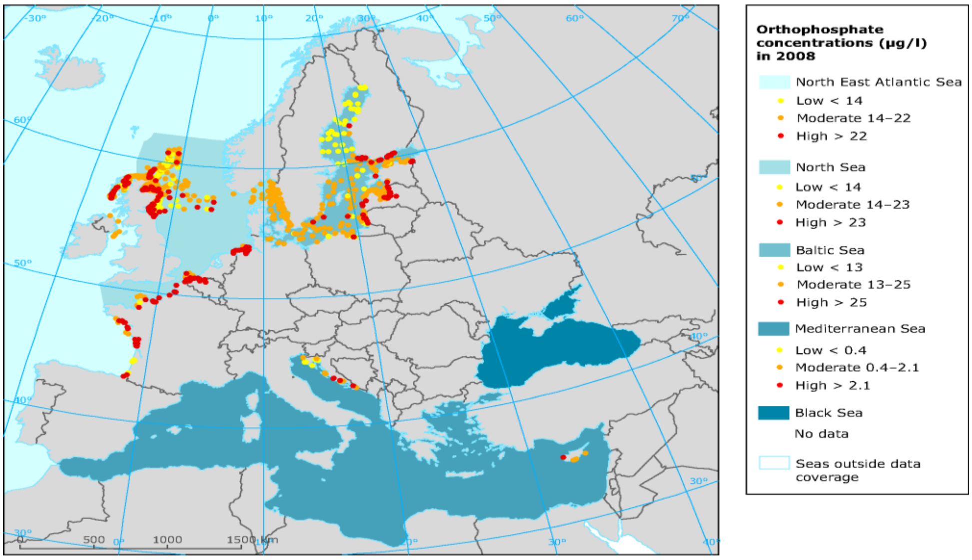


Figure 5-4 Orthophosphate concentrations in transitional, coastal and marine waters in 2008 (EEA, 2011)



The EEA (2005) noted that discharges of pollutants from point sources had decreased significantly over the past 30 years.

In **transitional, coastal and marine waters**, the EEA (EEA, 2011) found that in 2008 'the highest concentrations of oxidized nitrogen were found in the Gulf of Riga, and in Lithuanian, Swedish, German, Belgian, and Scottish coastal waters. Between 1985 and 2008, 12% of all the stations in the European seas reported to the EEA showed decreasing trends of oxidized nitrogen concentrations. These trends were more evident in the open Baltic Sea and in the Dutch and German coastal waters in the North Sea' (Figure 5-3). For orthophosphate, the highest concentrations 'were found at Finnish coastal stations in the Gulf of Finland, the Gulf of Riga, German, Belgian, French, and Scottish coastal waters. Between 1985 and 2008, 15% of all the stations in the European seas reported to the EEA showed a decrease in orthophosphate concentrations, mainly because of improved waste water treatment. This decrease was most evident in Norwegian, Lithuanian, Danish, Belgian and Dutch coastal water stations, and in the open waters of the North and Baltic Seas' (Figure 5-4).

The EEA (2010d) found that average nitrate concentrations in European **groundwater** increased from 1992 to 1998, and have remained relatively constant since then. The average nitrate concentration in European **rivers** decreased by approximately 9% between 1992 and 2008 (from 2.4 to 2.2 mg/l N) (Figure 5-5). Average orthophosphate concentrations in European rivers have decreased markedly over the last two decades, being almost halved between 1992 and 2008 (47% decrease). Also average **lake** phosphorus concentration decreased over the period 1992-2008 (by 26%), the major part of the decrease occurring in the first half of the period (Figure 5-6).

Figure 5-5 Nitrate concentrations in rivers between 1992 and 2008 in different geographical regions of Europe (EEA, 2010d)

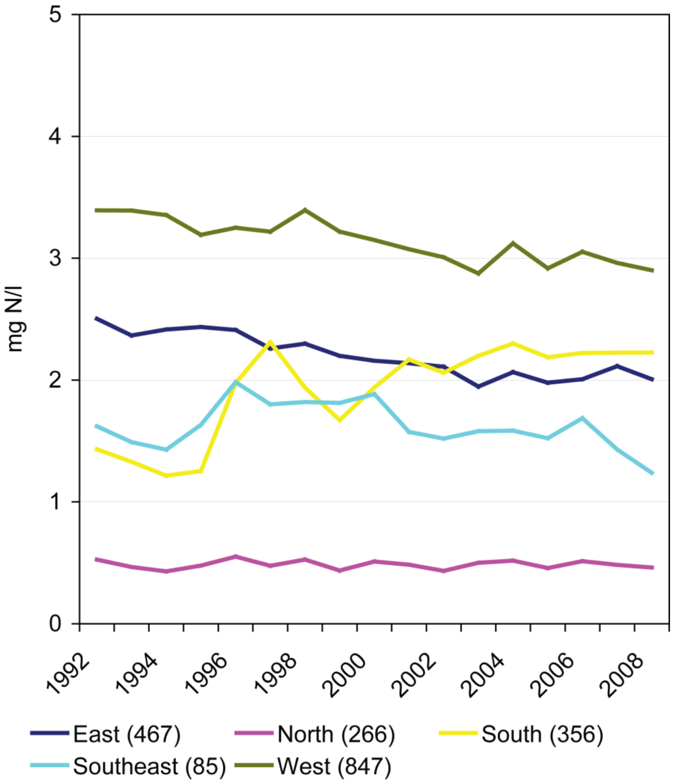
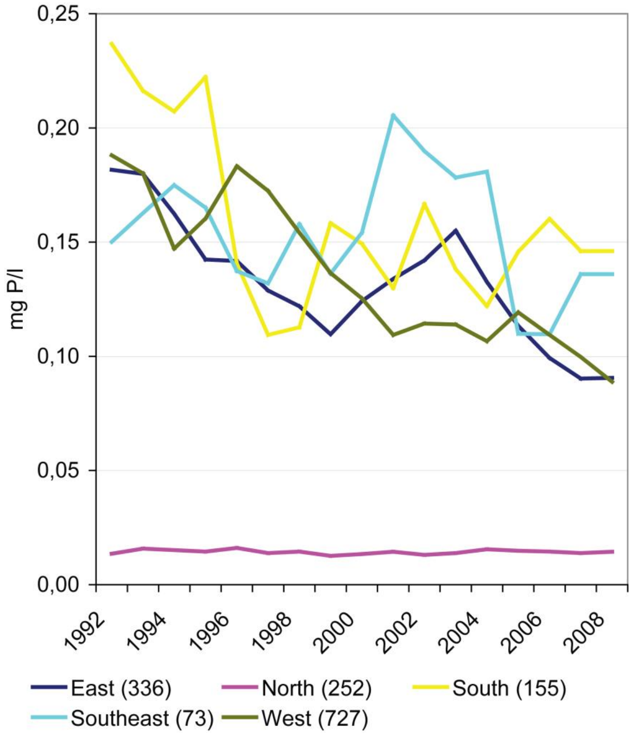


Figure 5-6 Phosphorus concentrations in rivers (orthophosphate) between 1992 and 2008 in different geographical regions of Europe (EEA, 2010d)



5.2.2 Measures to combat the pressure and their costs

Phosphorus removal from waste water can be achieved through reducing inputs (eg by improving detergent quality). However, given the input from human excreta, further removal is needed if there is a pressure on water. The two principle approaches are either chemical (precipitation of the phosphate using substances such as iron or aluminium and removal of the precipitate) or biological (using bacteria to incorporate the phosphate into cell biomass, which is subsequently removed as sludge).

Nitrogen is removed in waste-water treatment plants (WWTP) through a biological process whereby the nitrogen compounds (ammonia) are oxidised to nitrate. A second process then results in denitrification which causes the nitrogen to be released to the atmosphere as nitrogen gas.

The requirements for application of phosphorus or nitrogen removal are driven by the 1991 Urban Waste Water Treatment Directive[24] (UWWTD). The overall costs are addressed in the following section.

5.2.3 Costs of reducing pressures

Under the UWWTD waters at risk of eutrophication (Sensitive Areas) need to be subject to more stringent treatment (phosphorus and/or nitrogen removal). The UWWTD requires either that all discharges from agglomerations of more than 10,000 population equivalent (p.e.) meet the criteria for more stringent treatment (Article 5(2) and (3)) - or alternatively to ensure that the minimum percentage of reduction of the overall load leaving all urban WWTPs in that area is at least 75% for total phosphorus and at least 75% for total nitrogen (Article 5(4)).

The latest European Commission implementation report of the UWWTD (SEC(2011) 1561) noted that the identification and the designation of sensitive areas has been done in different ways across the EU:

- 12 Member States have considered their entire territory as a sensitive area or as a relevant catchment of a sensitive area, and therefore apply Article 5(8) of the Directive: AT, CZ, DE, DK, EE, FIN, LT, LU, LV, NL, PL, RO. Out of those, 8 Member States also apply Article 5(2) and 5(3), while AT, DE, NL and PL apply Article 5(4).
- 3 Member States, BE, SK and SE, apply Article 5(2) and 5(3) and have identified all their water bodies as sensitive areas.
- The remaining 12 Member States have identified particular water bodies in their territory as sensitive areas or catchment of sensitive areas, and therefore apply Article 5(2) and 5(3) of the Directive: BG, CY, ES, FR, EL, HU, IE, IT, MT, PT, SI and UK.

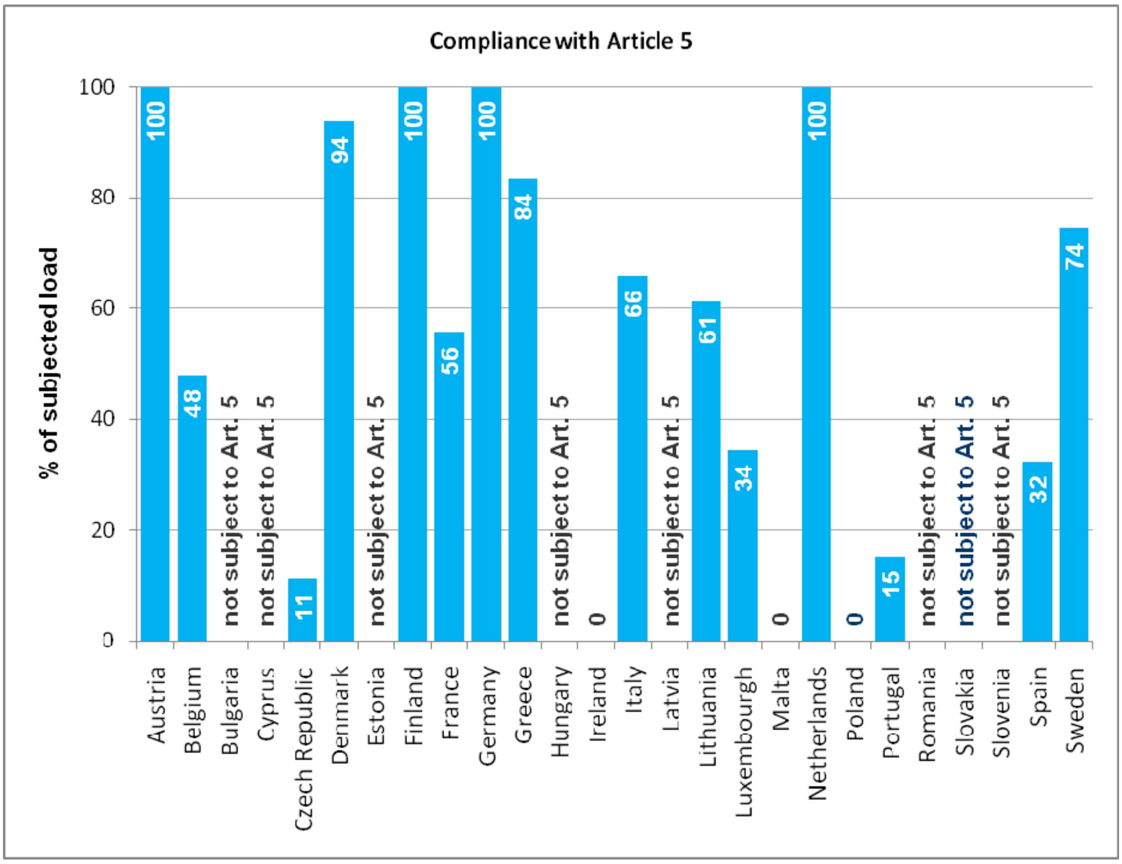
The costs of completing the implementation of the UWWTD represent the difference between current levels of compliance and the necessary investments to achieve full

[24] Directive 91/271/EEC concerning urban waste water treatment. OJ L135.

compliance. Figure 5-7 provides the levels of compliance with Article 5 from the 2011 Commission implementation report.

Figure 5-7 Assessment of compliance with Article 5 (in relation to the generated load subject to compliance with Article 5) for 18 Member State subject to compliance

Note that Member States that joined in 2004 and 2007 were given additional time for implementation and, therefore, were not yet "subject to compliance" under Article 5.



The costs of achieving compliance have been analysed by COWI (COWI, 2010) using data based on status at either the end of 2005 or 2006. The investments required for new investments are: Article 3 (waste water collection) €28,312 million; Article 4 (base treatment requirements) €4,495 million; and Article 5 (treatment in Sensitive Areas) €12,455 million; giving a total of €45,262 million. The compliance costs estimates do not take the actual state of the infrastructure into account. Thus there will also be further needs for annual re-investments to maintain and operate the WWTPs.

The Impact Assessment accompanying the Commission proposal for revising the detergents Regulation (SEC(2010) 1277, 4.11.2010) reported UK data stating that removal costs for phosphorus from waste water treatment plants are about €30 per kg phosphorus for capital and operating costs combined. The Commission stated that, extrapolating to the EU as a whole, 'taking into account the present degree of connection to tertiary treatment varying between 20% and 90%, the cost for P removal in WWTP would lie somewhere in the range of €10 million to €86 million'.

At this stage it is assumed that implementation of the UWWTD would address the major nutrient point sources. However, there is on-going analysis (due to be completed later in 2012) examining the information on pressures identified in Member States' RBMPs under the WFD. This may identify that there are additional important point sources. At this stage, it is not possible to determine how important, if at all, these sources might be. Of course, these costs are driven by the WFD.

Given the wide scope of the UWWTD and its targets it is assumed here that point source water pollution pressures will be reduced sufficiently to ensure the maintenance f existing ecosystem quality and the restoration of more than 15% of impacted waters by 2020. Some additional, mostly localised actions may be required in some cases where actions are not foreseen as necessary to meet UWWTD targets. However, further water pollution reduction measures may be identified as necessary to achieve Good Ecological Status or Good Ecological Potential of water bodies under the WFD. As explained in the introductory section, the measures would more than meet Target 2, so that this target would not require significant additional expenditure on point source pollution measures to that already envisaged under existing EU policies and legislation.

A question remains whether removing (or reducing) the pressure from WWTPs through implementation of the UWWTD will allow for improvement in ecosystem quality or event prevent further deterioration. Within rivers improvement of discharge quality routinely results in water quality improvements. However, problems can arise where there is significant sediment accumulation, such as in lakes and in marine waters. Here nutrients can be stored and released through various chemical processes and/or forms of disturbance. Thus while the implementation of the UWWTD will bring significant improvement in ecosystem quality across many EU water bodies, historical pollution from WWTPs may remain a problem for some time in the future (and certainly beyond the Target 2 deadline).

5.3 Eutrophication of water bodies from diffuse pollution

5.3.1 Description of the pressure and proportion of habitats affected

Nutrients derived from diffuse sources (such as nutrient-rich runoff from agricultural land) have similar impacts to those from point sources. The extent of eutrophication and the relative contribution of different sources are addressed in the section on point sources (see above).

5.3.2 Measures to combat the pressure and their costs

Measures to protect waters from diffuse pollution are driven by EU Directives such as the Nitrates Directive[25], the Groundwater Directive[26], the Sewage Sludge Directive[27], and the WFD. Member States also use CAP cross-compliance regulations and agri-environment schemes to work alongside regulation to encourage land management practices that can

[25] Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. OJ L375.

[26] Directive 2006/118/EC on the protection of groundwater against pollution and deterioration. OJ L372.

[27] Directive 86/278/EEC on the protection of the environment and in particular of the soil, when sewage sludge is used in agriculture. OJ L181.

address issues of diffuse pollution from agriculture in particular. In some Member States, considerable investment is also made in the provision of advice to farmers and other land managers.

Nutrient emissions to water can be reduced using different measures controlling emissions or pathways of nutrient movement. A number of measures reducing emissions at the source, such as reductions in fertilizer use, reductions in livestock and reducing the nitrogen content of fodder. Note that these measures may also reduce emissions to air and so contribute to reducing deposited pollutant eutrophication (see section 5.5.2). Nutrient leaching can also be reduced by measures such as use of catch crops, buffer strips, wetlands, etc. It is also possible, in limited cases to mitigate impacts by bio-manipulation of some smaller water bodies through fish-stock management or even to extract nutrient loaded sediments. However, these measures are very expensive and often technically difficult, so have limited application.

Estimates of costs associated with the implementation of the WFD in theory should give a good indication of the scale of funding needed to meet a range of objectives associated with both water quality and water quantity. A study by Ecologic (Dworak et al, 2010), collated available management and capital cost data from draft RBMPs in 2009 where available from eight Member States (FR, IE, EE, BE, SL, HU, BG, RO), relating to:

- livestock oriented farming measures – improving manure storage, reducing nutrient content of animal feed, reducing livestock numbers etc;
- chemical fertiliser input reduction measures;
- pesticide measures – including investment in farm infrastructure as well as integrated pest management measures, use of different crop varieties etc;
- integrated farming measures, including precision farming and organic farming; and
- land use intervention measures including land conversion to forestry.

However, it was not clear from the data, what time frame the costs relate to or the area of land that they are intended to cover. In the majority of cases there was also no information given about the intended source of the funding. Therefore as part of the IEEP Rural Land Management Costs study, see section 2.8, (Hart et al, 2011), a review of a number of final RBMPs was carried out to see if it were possible to ascertain any additional estimates of the costs of undertaking the programmes of measures (PoMs). The exercise revealed that where cost estimates are available, these are extremely variable in terms of the type of costs identified which makes any comparability very problematic. Nevertheless, the results indicated in Table 5-1 and Table 5-2 do give some indication of the scale of costs involved in meeting WFD requirements in different parts of Europe.

Table 5-1 Total costs of undertaking Programmes of Measures in selected River Basin Management Plans

Member State and RBMP	Total Costs	'Additional costs' as a result of WFD requirements
Netherlands	until 2015: €2.3 billion 2015–2027: €1.9 billion	
UK – South West	66 million £/yr	3 million £/yr
UK – Anglian	114 million £/yr	64 million £/yr
UK – South East	39.5 million £/yr	
Estonia – East Estonia	4030 million EEK/yr	330 million EEK/yr
Belgium – Scheldt	171-845 million €/yr	32—345 million €/yr
Spain: Catalunya	€6.3 billion to 2015	€4.2 billion to 2015

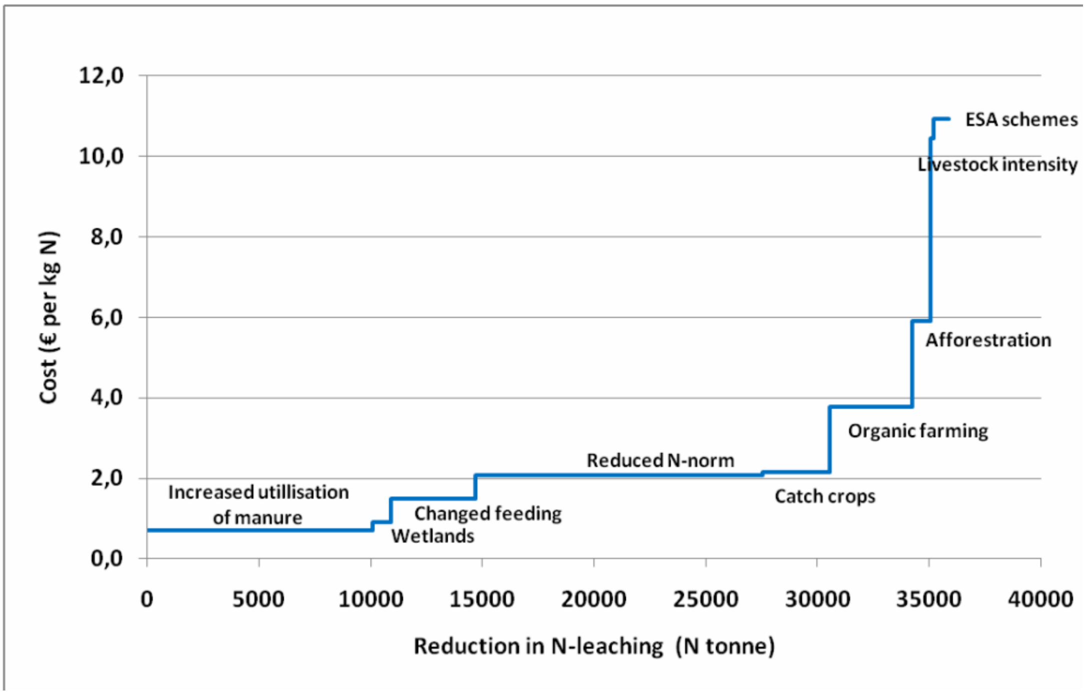
Source: (Hart et al, 2011) Selected River Basin Management Plans

Denmark has a particularly difficult problem regarding nitrate pollution (which resulted in that Member State being a major driver for adoption of the Nitrates Directive). Denmark has adopted Action Plans to reduce nitrogen pollution. Measures adopted have included wetlands, forestry, Environmental Sensitive Areas and organic farming. On farm measures included change in feeding practices, livestock density and utilization of nitrogen in animal manure.

The Commission's forthcoming review of RBMPs under the WFD has found that most RBMPs do not contain precise information on how much it will cost to implement the measures and how they are going to be financed. Therefore, expected new costing information from RBMPs is not forthcoming. Indeed, the Commission is critical of this lack of information to demonstrate that measures are to be financed (and, therefore, implemented) and also that costs are not analysed in comparison to benefits, leading to a lack of transparency on the justification for either action of inaction (such as use of exemptions).

In relation to specific measures, a study by Jacobsen (2004) examined the marginal costs of the different measures within the Action Plan II (Figure 5-8). This clearly shows that there are much lower marginal costs for some measures (better manure management, construction of wetlands and reducing nitrogen in feed) than for others (such as changing livestock intensity and ESA schemes).

Figure 5-8 Marginal cost curve (€ / kg N in reduced leaching) based on the implementation of Action Plan II for the Aquatic Environment in Denmark (Jacobsen, 2004)



Van den Broek et al. (2007) found that wet riparian buffer strips can decrease reduced nitrogen emissions to surface water by 15 kg N per ha and that this cost 37 €/kg N. Dry buffer strips are even more expensive at about 40 €/kg N. Van der Bolt et al. (van der Bolt et al, 2008) stated that average additional cost of reducing nitrogen emissions to surface waters in the Netherlands was 70 €/kg N for a group of measures including manure free zones.

Overall Kuik (2006) found that the estimated costs of the Nitrates Directive differ across Member States and that these range from 6 to 236 € per hectare affected, and from 0.4 to 3.5 € per kg N. However, it is important to note that these are costs associated with compliance with a legal objective. Cardenas et al (2011) in examining costs of implementing the Directive found that the costs of nitrate leaching mitigation measures, while effective, depend on local conditions. For dairy herds, a combination of measures (herd reduction, changed grazing, fertiliser and manure management) ranged from 197 to 800 € per hectare per year. For beef the average was 157 € per hectare per year. However, correct measures can save farmers money, such as correcting sulphur and potassium deficiencies in soil could save farmers around 106 € per hectare per year and better calculation of nitrogen requirements from fertiliser application could save 196 € per hectare per year.

A study by Gren et al. (2008) calculated the costs for different measures for countries around the Baltic Sea and found large variation in costs. The study included an assessment of the marginal costs of measures to control both point and diffuse source pollutants. Table 5-2 and Table 5-3 provide the marginal cost calculations. This is a very useful analysis given the wide differences in the starting points of the Member States, from Sweden with heavy pre-existing investment in a number of measures, to Poland where significant early investment is still needed.

The results show that there are significant differences between measures, between countries and between pollutants. Overall the marginal costs for most comparable measures are significantly higher for all countries for phosphorus than for nitrogen. For nitrogen, lower marginal costs for diffuse pollution controls depend upon the country, but changes of land-use, manure management and wetlands all have low marginal costs in some cases. For phosphorus, only use of P-free detergents has a significantly lower marginal cost for diffuse pollution (and prioritising point source reduction through sewage treatment is also more cost-effective). Other measures' marginal costs vary between countries and (given the ranges stated) vary between individual situations.

Bryhn (2009) also found that, for the Baltic, measures to remove total phosphorus such as abatement measures in agriculture and wetland construction had low cost-effectiveness compared to other options. Rather the most cost-effective measures identified were improved sewage treatment in the eastern part of the catchment, and banning phosphates in detergents. The author concluded that 'the total abatement cost for applying one or two of these measures to the extent that the trophic state would be restored to conditions prevailing before 1960 was estimated at 0.21–0.43 billion € per year'.

Measures may also lead to savings in other areas. For example, the Impact Assessment accompanying the Commission proposal for the revising detergents Regulation (SEC(2010) 1277, 4.11.2010) reported UK data that elimination of phosphates from detergents could lead to savings in operating costs at existing WWTPs in the order of €2.5 million per annum (note in the UK only 27% of the phosphorous is removed in tertiary treatment). If all waste water was subject to phosphorus removal, the savings across the UK would be between 8.9 and 16.5 million €/yr. The Commission stated 'extrapolation for the entire EU for a complete removal of 110,000 t P from detergents would lead to operational cost savings between €50 million and €96 million (assuming 100% connection to tertiary treatment).

Table 5-2 Marginal costs (€/kgN) for nitrogen reduction measures for countries surrounding the Baltic Sea (Gren et al, 2008)

Member State	Reducing nitrogen oxide emissions to air	Reducing livestock emissions	Reducing fertiliser use	Controlling sewage discharge	Reducing discharge from private sewers	Using catch crop to reduce emissions	Grass land to provide cover	Change of spread of manure	Energy forest	Wetlands
Denmark	243-387	540-613	0-1451	141-331	509-565	290-299	974-1007	75	1036-1071	66
Finland	246-396	282-552	0-394	141-419	509-717	146-317	313-317	52	822-1022	11-139
Germany	444-755	532-634	0-410	141-451	509-771	113-327	324-327	136	583-588	23-25
Poland	314-528	314-416	0-108	113-448	429-766	82-106	88-96	71	118-139	10-11
Sweden	218-373	214-493	0-472	141-745	509-766	46-376	75-302	44-71	544-3487	75-2764
Estonia	224-374	213-325	0-65	113-325	429-557	58-80	57-58	58	107-109	48-67
Lithuania	254-418	64-134	0-24	113-386	429-780	71	20	55	107	20
Latvia	228-351	208-407	0-156	113-456	429-660	139-204	139-140	66	280-283	64-93

Table 5-3 Marginal costs (€/kgP) for phosphorus reduction measures for countries surrounding the Baltic Sea (Gren et al, 2008)

Member State	Using P free detergents to reduce pollution	Reducing livestock emissions	Reducing fertiliser use	Controlling sewage discharge	Reducing discharge from private sewers	Using catch crop to reduce emissions	Using buffer strips to reduce ingress of pollution	Wetlands
Denmark	104-434	23796-45189	0-102649	573-1271	2397-2445	7455-9899	31067-37713	6992-8677
Finland	138-493	9561-16288	0-11215	573-1683	2397-3239	4933-5095	7419-8060	751-2363
Germany	255-1256	40451-56401	0-93506	573-3115	2397-5992	1444-1873	6716-8529	2999-3870
Poland	165-270	4673-5527	0-5180	385-1335	2013-3246	2941-3239	2941-3239	456-679
Sweden	102-941	11183-42687	0-38884	573-2350	2397-4522	7366-66649	6784-66649	25805-63801
Estonia	159-279	7271-8680	0-2613	385-1293	2013-3145	19055-88657	2377-2574	6158-9127
Lithuania	133-190	1120-1670	0-1503	385-1182	2013-2875	4206	891	2405
Latvia	170-340	4344-6106	0-2752	385-1380	2013-3355	8074-62177	2156-2675	4232-5119

5.3.3 Costs of reducing pressures

The preceding section provided information on the costs of individual measures. The overall costs of delivering Good Ecological Status or Good Ecological Potential of water bodies should be found in the RBMPs of Member States. However, this information, as found by the Commission, has not be determined (or made available). Thus the overall costs are not able to be determined, although, as demonstrated earlier, there would be no significant additional costs in achieving Target 2.

5.4 Nitrate pollution of ground-water

5.4.1 Description of the pressure and proportion of habitats affected

Groundwaters are often at risk from elevated nitrate levels. The Nitrates Directive views this issue as one of protection of drinking water sources and not, unlike surface waters, as one of eutrophication. Nitrates only affect biodiversity where the groundwater reaches the surface either into surface waters or to feed terrestrial habitats. For the former, this then becomes an issue of surface water quality (see above). For terrestrial habitats the impact is highly site specific.

The EEA (2005) found wide variation in groundwater nitrate levels in Europe, with low levels in Nordic countries, but relatively high levels in Western Europe. It also found that the general trend is ‘stable to increasing’, despite measures being adopted. The EEA, therefore, concluded that ‘is very difficult to prove a direct context between the application of nitrogen fertiliser in agriculture and the nitrate content in groundwater bodies as there is often a significant time lag between changes in agricultural practices and changes in nitrate concentrations in groundwater of up to 40 years, depending on the hydrogeological conditions’.

5.4.2 Measures to combat the pressure and their costs

The measures to be taken to reduce nitrate pollution to surface waters are effectively the same as to protect surface waters.

5.4.3 Costs of reducing pressures

The costs of reducing pressures using measures to be taken to reduce nitrate pollution to surface waters are effectively the same as to protect surface waters. However, it is not evident that there is any quantification of the extent of the biodiversity impact from nitrate in groundwater and, therefore, any possibility to estimate the extent of costs of reducing the pressure to any given target value.

The overall costs of delivering good chemical status of groundwater bodies should be found in the RBMPs of Member States. However, this information, as found by the Commission, has not be determined (or made available), although, as demonstrated earlier, there would be no additional costs in achieving Target 2.

5.4.4 Conclusions

There are still many critical pressures on Europe’s waters and the ecosystems that they contain and that they maintain. The recently published Blueprint to Safeguard Europe’s

Water Resources (COM(2012)673) describes these pressures and the difficulties faced in addressing them. Furthermore, it highlights the difficulties being faced by Member States in implementing the WFD to deliver its improvements in aquatic ecosystems.

However, while many challenges are understood, the legal framework to deliver ecosystem restoration and maintenance for Europe's waters is in place. The WFD requires the achievement of Good Ecological Status or Good Ecological Potential which are equivalent to restored ecosystems. Furthermore, it requires that the status of water bodies is not allowed to decline, thus being equivalent to the Target 2 objective for maintaining ecosystem quality.

The analysis of progress in implementing the WFD shows that, even with missing RBMPs from a few Member States, it is expected that more than 15% of water bodies in the EU not currently at good status will be restored to good status by 2020. Therefore, achieving the Target 2 restoration target will not result in any additional costs over and above that already required by the WFD adopted in 2000.

The WFD requirement to not allow for a deterioration in water body status seems to be consistent with the Target 2 objective to ensure maintenance of ecosystems. Implementation of this objective would also not result in any additional costs over and above that already required by the WFD.

Furthermore, many restoration costs occur through the implementation of older directives, such as the UWWTD and Nitrates Directive. These impose significant costs, but are delivering major improvements in water quality, contributing to WFD objectives.

5.5 Air pollution

5.5.1 Introduction

The following sections describe the pressures on biodiversity arising from eutrophication from deposited nitrogen from atmospheric pollution, deposited acidity and pressures from ground level ozone. Each section also describes the measures that can be taken to tackle these pressures.

Overview of targets and progress towards these

Firstly, it is important to understand the current predicted changes in air pollution impacts on ecosystems and their relationship to Target 2. Currently policy targets are driven by the 2005 Thematic Strategy on air pollution and EU air pollution legislation that supports this strategy. The Thematic Strategy objectives for ecosystems included the following changes for 2020 compared to 2000:

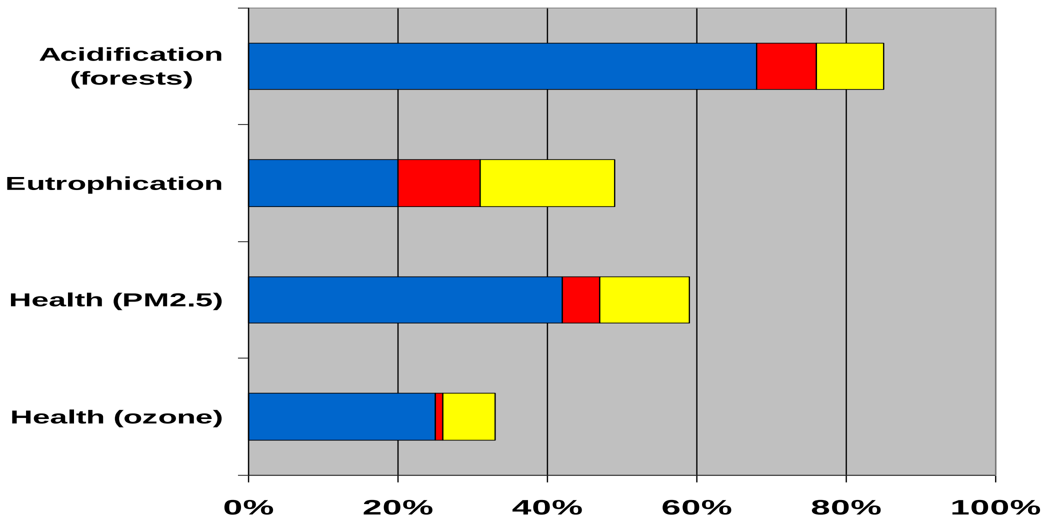
- 74% less forest area (63,000 km^2 compared to 243 km^2) and 39% less freshwater area (19,000 km^2 compared to 31,000 km^2) where acidification critical loads are exceeded.
- 43% less area where critical loads for ecosystem eutrophication are exceeded (416,000 km^2 compared to 733,000 km^2).

- 15% less forest area where critical levels are exceeded due to ozone (699,000 km^2 compared to 827,000 km^2).

The objectives would seem to be consistent with meeting the restoration target within Target 2 (if the reduction in pressure allows for ecosystems to then recover), but they would not ensure the maintenance of all sensitive ecosystems and, for these, further degradation would occur. The Thematic Strategy stated that these improvements would be delivered by reducing SO_2 emissions by 82%, NOx emissions by 60%, VOCs by 51% and ammonia by 27% relative to emissions in 2000.

Figure 5-9 sets out how progress towards meeting the targets of the Thematic Strategy for 2020 would be achieved. The blue bars are progress due to measures adopted before 2005, the red bars are cost-effectiveness measures and policies adopted in the Thematic Strategy and the yellow bars represent the maximum technical achievement potential in 2005 irrespective of cost-effectiveness.

Figure 5-9 Expected contributions of measures towards the achievement of targets under the Thematic Strategy for air pollution



It can be seen that pre-existing legislation was expected to deliver the majority of progress towards targets, supported by some additional measures. All of these objectives would be equal or greater than an improvement of 15%, but would not maintain all ecosystems as required under Target 2.

With regard to air pollution impacts, the removal of the threat of damage (eg no longer exceeding a critical load) is the pre-requisite to restoration. However, it should be stressed that reducing deposition of acidifying or eutrophicating substances to below the critical level for an ecosystem is **NOT** the equivalent of restoration. It only means that the continuing damage due to that deposition no longer occurs. Recovery of soils, for example, may take some time, let alone the recovery of specific species and communities representative of those ecosystems. However, whatever that restoration timescale, there would be technically no need for further reductions in deposition and, therefore, no need for further costs for air pollution abatement. Therefore, restoration of damaged ecosystems would need to be addressed through appropriate site management actions (if possible) as discussed in Chapter 6.

Current research (Amann, 2012) for the forthcoming review of EU air policy using the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS)-Model has concluded that existing policies will deliver the following:

- SO_2 emissions will fall drastically: significant changes in fuel consumption levels and patterns that are expected in the baseline projection, together with progressing implementation of emission control measures, will lead to a reduction of about 70% in SO_2 emissions. Most of the SO_2 decline will emerge until 2020, for which emissions are estimated to be 68% below their 2005 levels.
- Overall, NOx emissions are expected to decline by more than 65% until 2030. This decline will happen gradually as a consequence of the staged introduction of more stringent emission controls to new vehicles and plants; in 2020, NOx emissions should already be 50% below the 2005 levels.
- Only minor changes are expected for NH3 emissions. While emissions declined between 2000 and 2010, they are expected to return to the 2005 level by 2030.
- VOC emissions will decline by 40% up to 2030, with the largest decline before 2020.

Recent work (EEA, 2012a) on the impacts of air pollution has also shown that between 2010 and 2020 the percentage of the EU-27 where critical loads for acidification will be exceeded will reduce from 11.35% of the EU area to 9.07% (20% of exceeded area improving) based solely on implementing current legislation. However, for nitrogen eutrophication the area exceeded will reduce from 68.7% to 64.0% (7% of exceeded area improving) based solely on implementing current legislation.

These results show that current emission reduction predictions are not sufficient to meet some of the Thematic Strategy expected emission reductions or the need to maintain ecosystems under Target 2 (although they would meet the 15% restoration component of Target 2 with the exception of ozone reduction). The modelling work (Amann, 2012) further explores a 'Maximum Technically Feasible Reduction' (MTFR) scenario explores to what extent emissions could be further reduced beyond what is required by current legislation, "through full application of the available technical measures, without changes in the energy structures and without behavioural changes of consumers". This indicated a further reduction in NOx emissions of 15-20% compared to baseline in 2020, a 30% reduction in NH3 and a 35% reduction in VOCs. The EEA study (2012a) shows that the MTRF scenario would reduce the area of the EU-27 where eutrophication is exceeded to 28% in 2020 – a reduction in exceeded area from 2010 of 59%.

Furthermore, the economic component of the GAINS model found that the costs of the current legislation air pollution control measures would increase from 0.4% of GDP in 2005 to 0.6% in 2010 (more than half due to EURO emission standards for road vehicles[28]). After 2020, costs are expected to reduce to 0.4% of GDP in 2050. However, full implementation of all available technical emission control measures would increase costs by 50%. However, the research found that costs could be 20% lower if measures were taken to control carbon emissions as these would have a knock-on benefit for air emissions. Therefore, climate and air pollution policies are strongly inter-related.

Considering interactions between pollutants

It is not possible to set out the costs of pollution control separately for these three pressures. This is primarily because the pressures strongly interact with regard to the effects of the individual pollutants. Deposited nitrogen arises from emissions of nitrogen oxides and ammonia, deposited acidity arises from emissions of sulphur dioxide, nitrogen oxides and (from subsequent denitrification) ammonia and ozone is a secondary pollutant driven by emissions of nitrogen oxides and volatile organic compounds (VOCs). Therefore, controls on a number of pollutants contribute to reducing more than one pressure.

It is important to stress that extensive work has been undertaken for many years examining cost-effective options to address air pollution, analysing different impacts, different pollutants and different pollutant sources. Furthermore, these analyses have not just focused on the costs and benefits of controls to reduce impacts on the natural environment, but also on human health, which provides additional strong benefits to actions taken to protect biodiversity. The benefits to health are not included in this report. However, it is important to stress that these arise in controlling these pollutants.

Therefore, the assessment of the costs of emission control is not considered separately in each of the following sections, as this makes no practical sense. Instead a final section presents cost analysis for the control of air pollutants as a whole.

A further important overarching consideration is the policy context for consideration of future emission reductions. Overall Member State emission reductions have been driven by the National Emission Ceilings Directive[29] (NECD) (covering all of the above pollutants), the targets for which were supposed to be met by 2010 (although several Member States missed those targets). The Integrated Pollution Prevention and Control[30] (IPPC) and Industrial Emissions[31] Directives are expected to continue to drive down a number of emissions as is continued implementation of controls (vehicle emission standards and fuel quality standards) for transport. Finally, the Air Framework Directive, in meeting ambient air limit values, is an additional pressure on emissions. All of these policies are framed by the 2005 Thematic Strategy on Air Pollution (COM(2005)446), which established overarching goals.

[28] See: Regulation on type approval of motor vehicles and engines with respect to emissions from heavy duty vehicles (Euro VI) and on access to vehicle repair and maintenance information amending Regulation (EC) No 715/2007 and Directive 80/1269/EEC, 2005/55/EC and 2005/78/EC.

[29] Directive 2001/80/EC on national emission ceilings for certain atmospheric pollutants. OJ L309.

[30] Directive 2008/1/EC of the European Parliament and of the Council of 15 January 2008 concerning integrated pollution prevention and control (Codified version). OJ L 24.

[31] Directive 2010/75/EU on industrial emissions (integrated pollution prevention and control). OJ L334.

Therefore, the 'baseline' for this study is not static and is different from the baselines used in the modelling of cost-effectiveness for air pollution control.

5.5.2 Eutrophication from air-borne nitrogen deposition

Description of the pressure and proportion of habitats affected

Deposited atmospheric nitrogen impacts on biodiversity through a range of processes. Species and communities most sensitive to this pollutant are those that are adapted to low nutrient levels. It is also important to note that gaseous ammonia can be harmful to vegetation, especially bryophytes and lichens, through direct foliar damage.

Dise (2011) provides a comprehensive review of the impacts of deposited nitrogen on ecosystems in Europe. Many semi-natural grassland communities in Europe are characterised by low nitrogen levels and are dominated by species with low nutrient requirements. Montane, boreal, tundra, subarctic, and arctic habitats are also vulnerable to nitrogen deposition, particularly the bryophyte and lichen dominated communities. Heathland communities are highly sensitive to nitrogen deposition due to their nutrient-poor acidic soils and, although across Europe a number of these communities are in low nitrogen deposition areas, they are at particular risk in some areas (eg the Netherlands). Wetland communities vary in their sensitivity to atmospheric nitrogen deposition. Nutrient poor bogs and fens are susceptible. The occurrence of these habitats in areas of northern Europe where nitrogen deposition is elevated makes these areas at risk. Forests on nutrient poor soils across Europe can be highly vulnerable to nitrogen deposition, with changes in biomass and ground flora composition. Shrub communities of the Mediterranean region are less well understood, but there is evidence of sensitivity to nitrogen deposition. A business-as-usual scenario estimates that by 2050 approximately 69% of the Mediterranean region will have nitrogen deposition exceeding 10 kg N/ha/yr (which is widely considered as a threshold for nitrogen impacts) (Bobbink et al 2003 and Phoenix et al 2006 quoted in (Dise, 2011).

Modelling to support future air pollution policy development (Amann, 2012) provides an assessment of the extent of the problem with regard to ecosystem area with nitrogen deposition exceeding critical loads. It concludes that the target for the Thematic Strategy on air pollution would require (for 2020) a reduction of 31% compared to 2000 levels (1188.4 X 1000km2 in 2000 compared to 820.0 X 1000km2 in 2020). The PRIMES model (baseline of current policies) shows that in 2020 the area that critical loads exceeded would be 950.3 X 1000km2 in 2020. Applying the MTFR in the EU-27 would reduce the area that critical loads are exceeded to 640.8 X 1000km2 in 2020.

The EEA (2012a) has recently assessed the extent of threats to habitats from nitrogen deposition and how this might change. This has been calculated as the share of sensitive ecosystems for which deposition of oxidized and reduced nitrogen compounds exceeds the critical load.

As shown in Figure 5-10 in 13 EEA member countries the percentage of sensitive ecosystem area at risk is close to 100% for 2010 and in only five EEA member countries is the area at risk less than 50% (Figure 5-11). Furthermore, Figure 5-12 shows that no significant improvements are predicted for 2020 in almost all EEA countries through the implementation of current policies. However, analysis does show that maximum application of all feasible technical measures (MTFR scenario) would produce significant improvements, but major threats would remain. In particular, while the improvements would deliver a restoration of more than 15%, it would not eliminate the pressures on ecosystems in some areas and, therefore, the maintenance objective for ecosystems within Target 2 would not be achieved.

Figure 5-10 Exceedance of critical loads for eutrophication due to the deposition of nutrient nitrogen in 2010. Source: (EEA, 2012a)

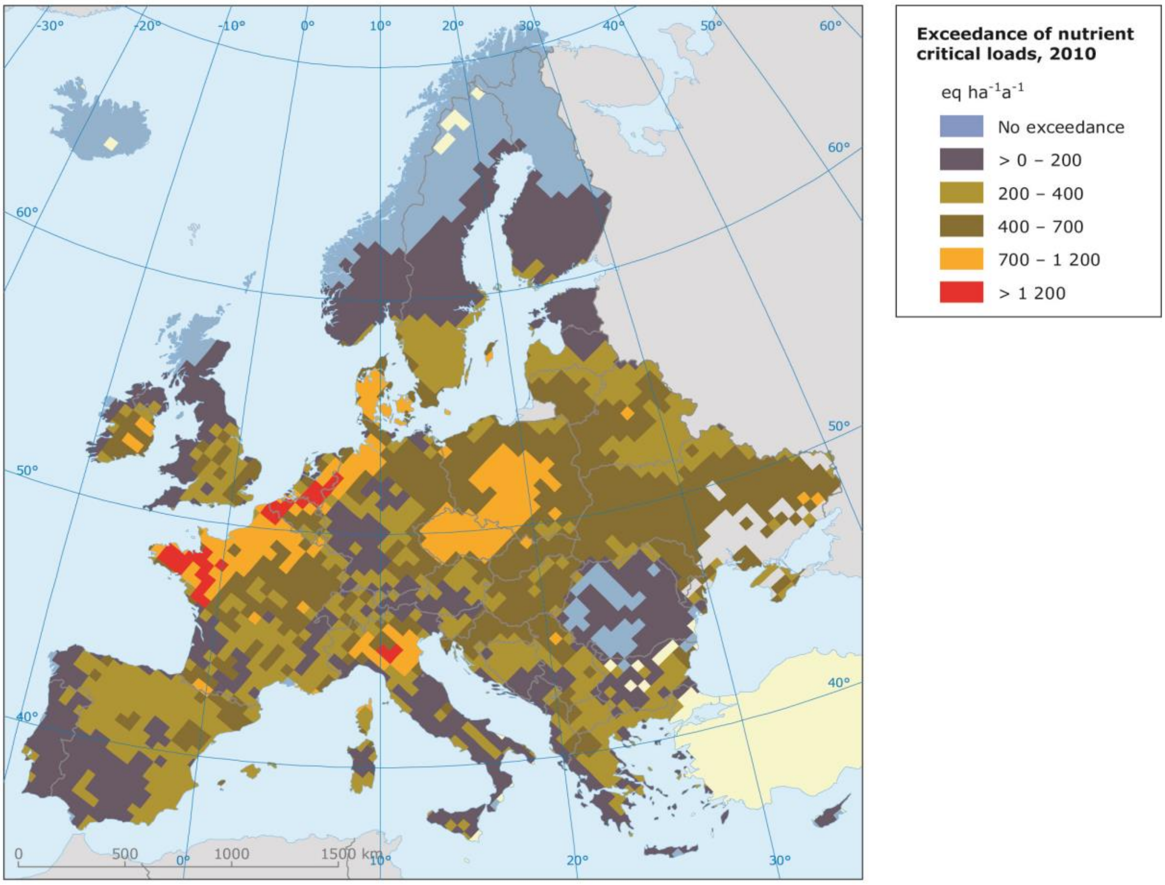


Figure 5-11 Percentage of ecosystem area at risk of eutrophication for EEA Member Countries and EEA Cooperating Countries in 2010 for a current legislation scenario. Source: (EEA, 2012a)

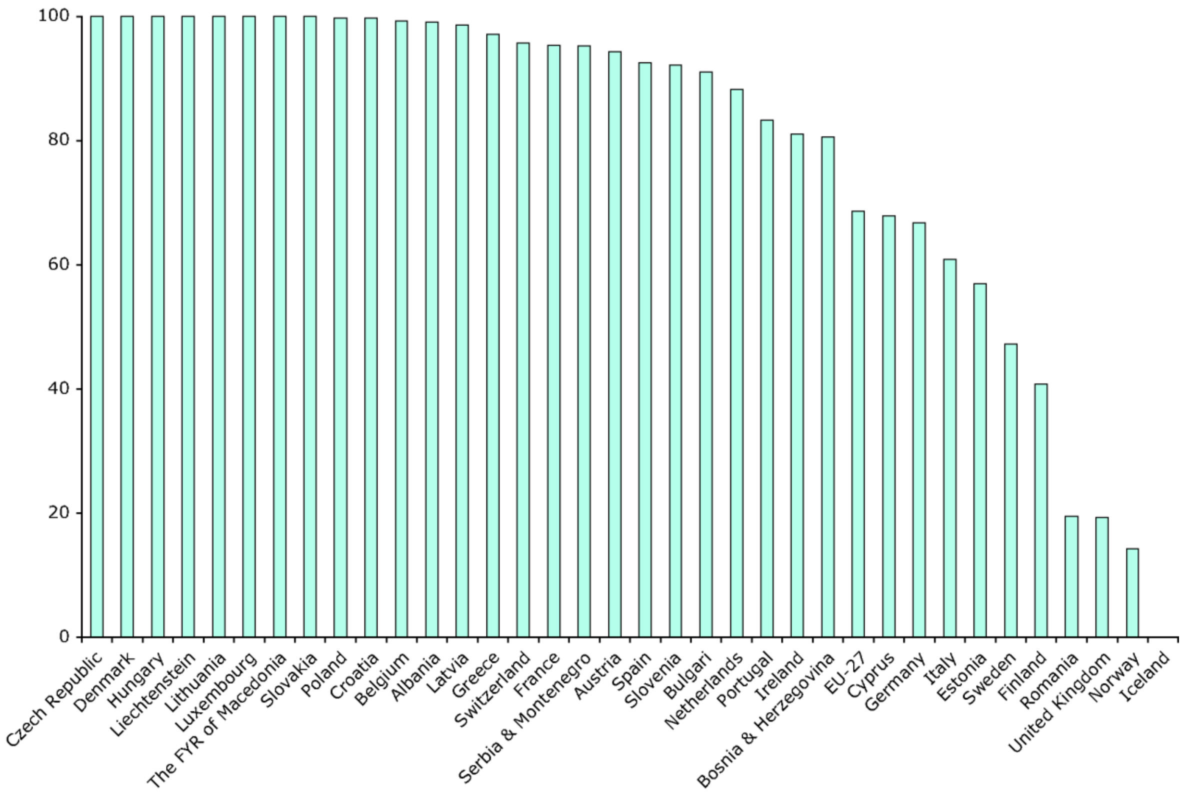
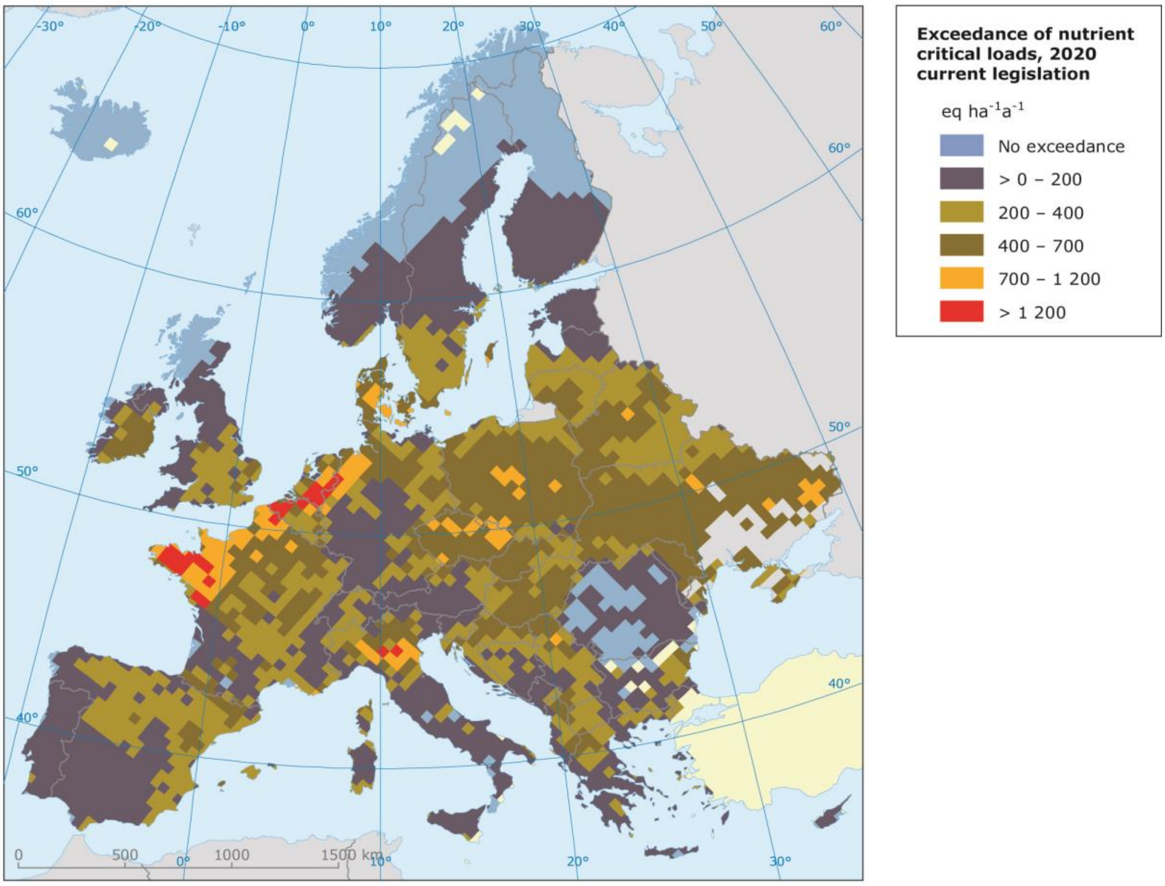


Figure 5-12 Exceedance of critical loads for eutrophication due to the deposition of nutrient nitrogen in 2020 under current legislation to reduce national emissions. Source: (EEA, 2012a)



Measures to combat the pressure and their costs

The main source of nitrogen oxides is combustion processes (industry, energy and transport). The nitrogen component (unlike) sulphur is not a constituent of the fuel, but the source is the air used in the combustion. Ammonia is also a product of some industrial activities, but its main source is agriculture, in particular animal manure.

Measures to control ammonia include low nitrogen feed, low emission housing for cattle, pigs, and poultry, air scrubbing, covered slurry storage, low ammonia application of slurry and solid manures, incineration of poultry manure, and urea substitution. These options often are of limited use on their own, as retention of nitrogen in housing can be subsequently emitted as manure is applied to land. Thus an integrated approach is needed.

Nitrogen emissions from combustion can be reduced by modifying the combustion conditions (eg low NOx burners) or treatment of the flue gases (by selective catalytic reduction (SCR) or selective non-catalytic reduction (SNCR)). For transport measures include changes in engine design, fuel quality, treatment of exhaust gas by catalytic converters, on-board diagnostic systems, etc.). These measures can reduce emissions from combustion processes by 50%–60% and by over 90% for catalytic converters.

There is also a wide range of non-technical measures used to reduce nitrogen emissions. City planning, provision of public transport and other forms of traffic management reduce nitrogen oxide emissions. Within the agriculture sector ploughing techniques, land management changes, etc., all can reduce ammonia emissions.

Given the wide range of potential sources and measures to control these, the costs of individual options vary significantly. A study (Oltmer et al, 2009) examined the costs of possible measures to reduce nitrogen emissions. Costs of technical measures range from 3.5 M€ for changing manure processing across Europe (resulting in 140 mol/ha/y emission reduction), to 6.8 €/animal place for placing air scrubbers in pig stables (resulting in 8 mol/y emission reduction). Moving farms may cost 2-7 € per avoided mol, while adjusting stables may cost 59M€/y resulting in 25 mol/ha/y emission reduction.

Capital costs to retrofit SCR NOx are 300-350 $/kW. Operating costs include new catalyst at a cost of 4,000-5,000 $/cubic meter and ammonia-based reagent at 400-600 $/ton (Cichanowicz, 2010).

Given the very large number of possible measures to control nitrogen oxides, it is not possible to elaborate their individual costs further here. A UK study (DEFRA, 2007) reviewed the costs for many different types of control and these vary. Which measures are most appropriate will also vary between Member States.

Costs of reducing pressures

The costs of reducing pressures are addressed in combination with other air pollutants in a subsequent section.

5.5.3 Acidification from air-borne sulphur-dioxide

Description of the pressure and proportion of habitats affected

Anthropogenic sulphur dioxide emissions are due largely to the combustion of sulphur-containing fuels (oil and coal) used in power stations, other stationary combustion activities and process industries (refineries). Nitrogen oxides are emitted by combustion processes; of which transport, power generation and heating are the most important sources. The sensitivity of ecosystems to acid deposition is related to soils or waters with limited or no acid neutralising capacity. These include oligotrophic waters, peatlands, heathlands, many forest soils as well as specific communities such as epiphytes.

European ecosystems are sensitive to the deposition of sulphur and nitrogen. Sulphur dioxide and nitrogen oxides exert both a direct and an indirect influence on organisms and materials. In the Nordic countries, acidification of soils and surface waters is attributed mainly to sulphur deposition. However, nitrogen is estimated to contribute 10 to 30% of the acidification in Denmark, southern Sweden and southern Norway. In some parts of Central Europe, nitrogen is responsible for a major part of acidification, and it seems likely that it will increase in importance as a source of acidification. This is explained by more stringent reduction of sulphur emissions as a consequence of international protocols, while nitrogen emissions seem to be continuously increasing due to insufficient measures to achieve reductions, which are largely counteracted by a relentless expansion in the numbers and use of the motor car.

Modelling to support future air pollution policy development (Amann, 2012) provides an assessment of the extent of the problem with regard to acidification of forest soil critical loads. It concludes that the target for the Thematic Strategy on air pollution would require (from 2020) a reduction of 74% compared to 2000 levels (208.3 x 1000 km^2 in 2000 compared to 72.9 x 1000 km^2 in 2020). The PRIMES model (baseline of current policies) shows that in 2020 the area that critical loads exceeded would be 91.2 x 1000 km^2 in 2020. Applying the maximum technically feasible reductions in the EU-27 would reduce the area that critical loads are exceeded to 45.2 x 1000 km^2 in 2020, ie a 78% reduction compared to 2000 levels. However, despite this reduction, 22% of sensitive ecosystems would remain affected by deposition levels above critical loads.

With regard to acidification of surface waters the modelling (Amann, 2012) concludes that the target for the Thematic Strategy on air pollution would require (form 2020) a reduction of 39% compared to 2000 levels (54.3 x 1000km^2 in 2000 compared to 32.6 x 1000 km^2 in 2020). The PRIMES model (baseline of current policies) shows that in 2020 the area that critical loads exceeded would be 21.7 x 1000 km^2 in 2020. Applying the maximum technically feasible reductions in the EU-27 would reduce the area that critical loads are exceeded to 16.1 x 1000 km^2 in 2020, ie a 71.4% reduction compared to 2000 levels. However, despite this reduction, 29.6% of sensitive ecosystems would remain affected by deposition levels above critical loads.

Therefore, for both soils and freshwaters the reduction in deposition would reduce the pressure on more than 15% of currently affected ecosystems. In many cases this might not lead to an immediate 'restoration', but is a precondition to such restoration. However, the potential reductions in deposition would not remove the pressure and would not, therefore, ensure the maintenance of ecosystem quality across Europe as required by Target 2.

Figure 5-13, Figure 5-14 and Figure 5-15 show that in 2000 large areas with exceedances (ie higher than 1 200 eq $ha^{-1}a^{-1}$, shaded red) are mostly located in Belgium, Germany, the Netherlands and Poland. With current legislation the size of the area where critical loads are exceeded is considerably reduced in 2020. However, using the maximum feasible reduction in emissions, many areas in Europe will no longer be at risk of acidification in 2020.

Figure 5-13 Exceedance of critical loads for acidification by deposition of nitrogen and sulphur compounds in 2010. Source: (EEA, 2012a)

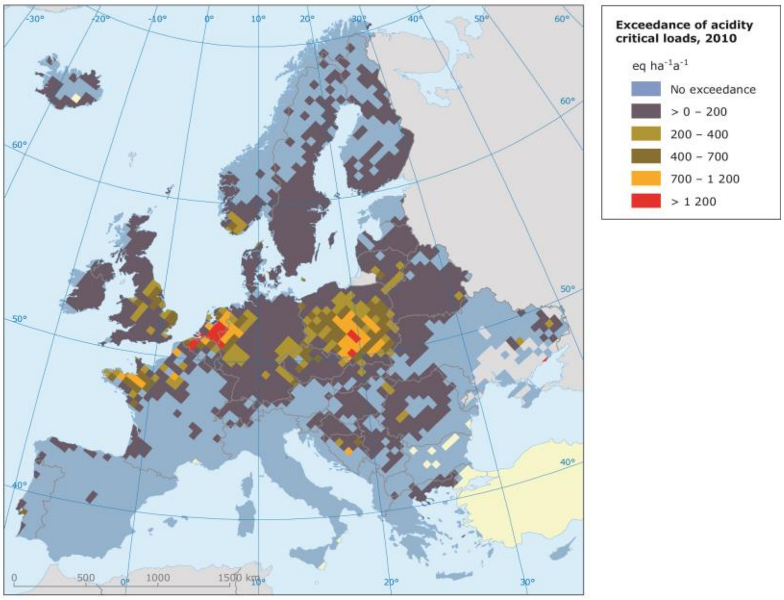


Figure 5-14 Exceedance of critical loads for acidification by deposition of nitrogen and sulphur compounds in 2020 under current legislation to reduce national emissions. Source: (EEA, 2012a)

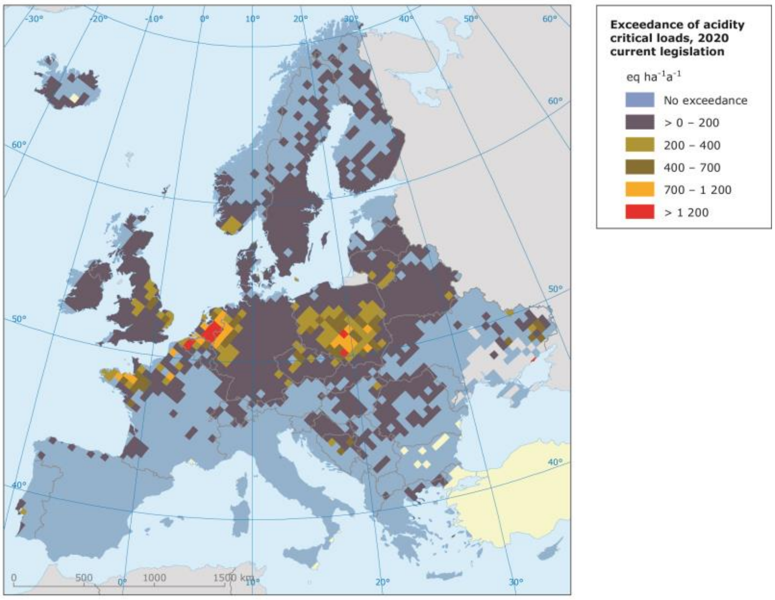
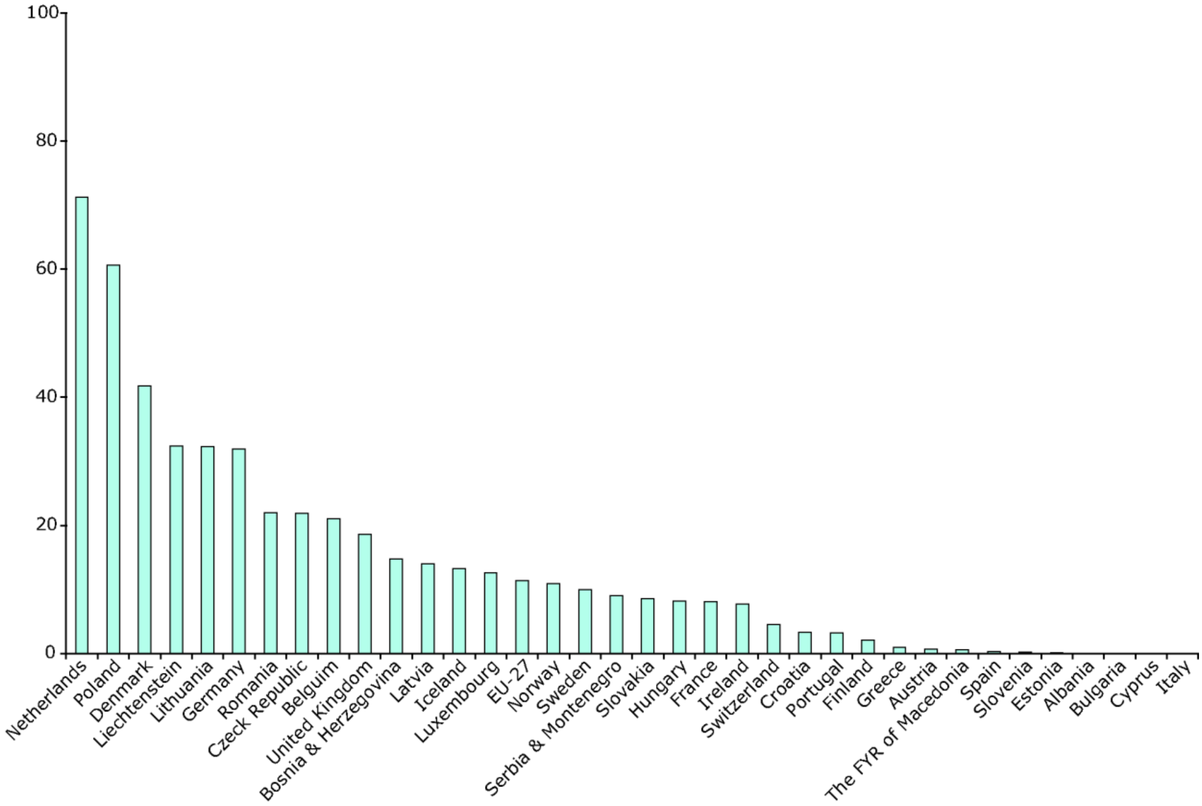


Figure 5-15 Percentage of ecosystem area at risk of acidification for EEA Member Countries and EEA Cooperating Countries in 2010 for a current legislation (CLE) scenario. Source: (EEA, 2012a)



Measures to combat the pressure and their costs

A range of measures can be used to reduce acid emissions. Those relating to nitrogen oxide are included in the previous section. To reduce emissions of sulphur dioxide, there are four principle alternatives:

- end-of-pipe solutions for combustion processes, such as flue gas desulphurisation;
- alteration of the combustion process, such as fluidised bed combustion;
- change in the fuel type, such as low sulphur fuel for diesel and shipping or low sulphur coal for combustion processes; and
- use of alternatives, such as reducing energy needs, changing energy sources, etc., so avoiding the emissions.

All of these measures are driven by EU policy currently. Fuel quality is mandated in EU law and the IPPC/Industrial Emissions Directive requiring ongoing improvements in combustion processes through the application of Best Available Techniques. Costs arising from implementation of this legislation in the future are, therefore, already required.

The capital cost to retrofit wet flue-gas desulphurization to a 500 MW combustion process was about 407 $/kW in 2008-2010 (Cichanowicz, 2010).

In its proposal to reduce the sulphur content of maritime fuels from 1.5% to 0.1% in sensitive areas such as the Baltic Sea in 2015 and from 4.5% to 0.5% in 2020 for all other areas (COM(2011)439), the Commission estimated the expected cost to the shipping industry of the new standards at between €2.6 billion and 11 billion, while the Commission noted that the benefits to health alone would be €34 billion.

Costs of reducing pressures

The costs of reducing pressures are addressed in combination with other air pollutants in a subsequent section.

5.5.4 Ozone pollution

Description of the pressure and proportion of habitats affected

Ozone is a secondary pollutant produced by the interaction between nitrogen dioxide and a range of volatile organic compounds in the presence of sunlight. Therefore, its presence depends on the relative availability of the primary pollutants and climate conditions.

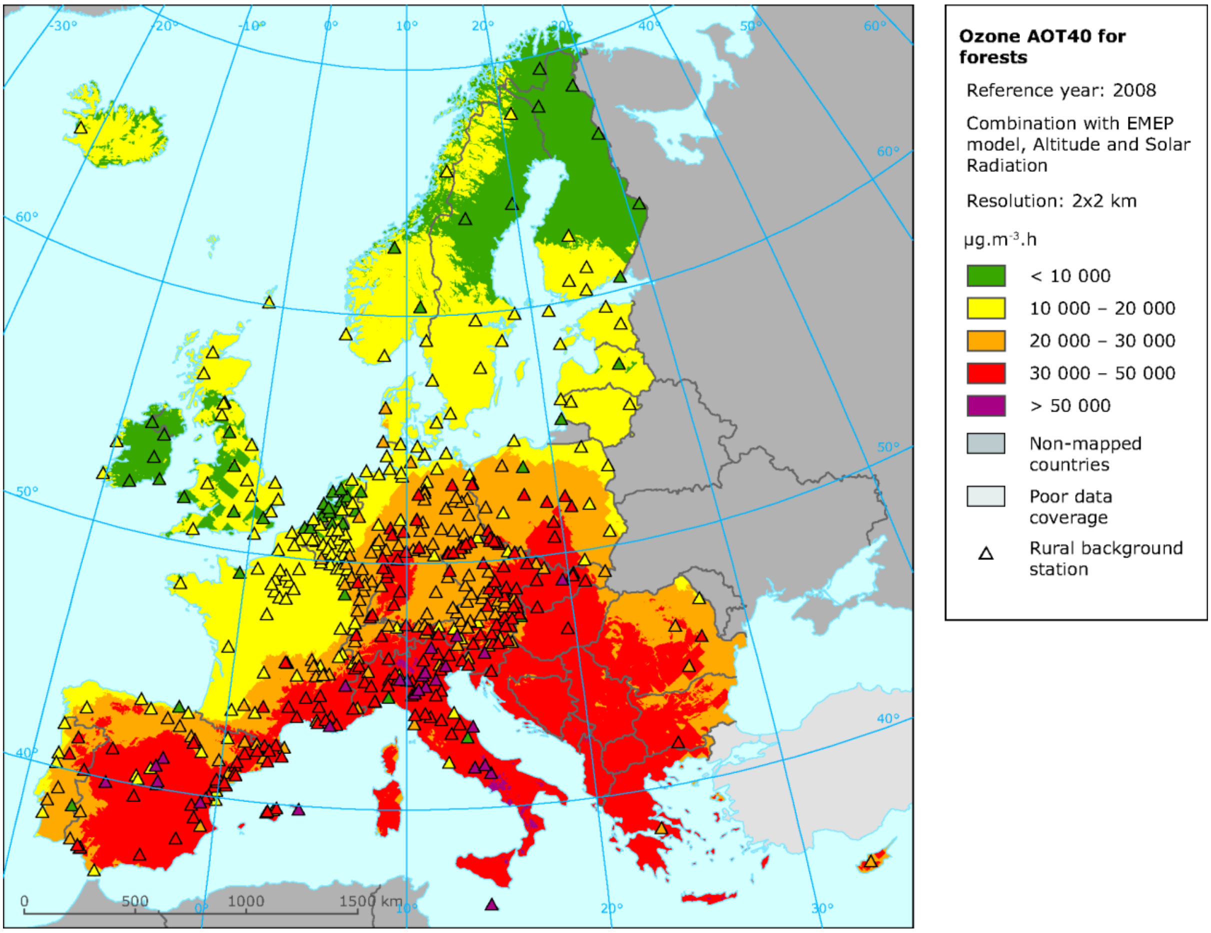
Ozone is a strong oxidising chemical that damages vegetation. However, the impact is not simply a factor of individual concentrations, but of an overall dose over a season. Therefore, the assessment of dose is determined by the cumulative concentrations above a particular threshold, known as the AOT40 or as a cumulative concentration. For example, the EU has the objective of protecting vegetation from ozone accumulated over the growing season (May to July). The target value for 2010 is 18,000 $\mu g/m^3$/hour. The long term objective is 6,000 $\mu g/m^3$/hour.

Assessment of the extent of the ozone problem focuses on the extent of exceedence of targets for agricultural or rural areas (given the potential impact on crops), although the problem would apply to important habitats in the same areas. There is also a specific focus on assessment of the impact on forests.

The EEA (2012a) states that in 2008 in the EEA-32 countries, about 35% of a total agricultural area (2052 million km^2) exceeded the EU target value. Exceedances of the target values are most notable in southern, central and eastern Europe. The long-term objective is met in 5% of the total agricultural area, mainly in Ireland and Scandinavia.

Within the UNECE Convention on Long-range Transboundary Air Pollution a critical level has been defined for the protection of forests - the accumulated sum during the summer (April-September) at 10,000 µg/m3/h. The gradient of the AOT40f values is relative low in northern Europe, and the highest values observed in the countries around the Mediterranean (Figure 5-16). The critical level is met in north Scandinavia, Ireland, part of the UK and in the coastal regions of the Netherlands (total forested area with concentrations below the critical level is 22% of a total area of 1.44 million km^2). In south Europe levels may be as high as 4-5 times above the critical level.

Figure 5-16 Rural concentration map of the ozone indicator AOT40 for forest in 2008 (Source: (EEA, 2012a)



Measures to combat the pressure and their costs

To reduce the impact of ozone pollution, measures need to be taken to control the two primary pollutants – nitrogen dioxide and VOCs. Information about measures to control the former is given in the previous section on eutrophication.

VOCs are produced by a wide range of processes – almost all processes that use solvents. They are regulated by the Solvent Emissions Directive[32] and IPPC Directive (now combined in the Industrial Emissions Directive), the VOC (petrol) Directive[33] and the Paints Directive[34]. Measures include substitution (eg water based paints for solvent based), simple enclosure

[32] Directive 1999/13/EC of 11 March 1999 on the limitation of emissions of volatile organic compounds due to the use of organic solvents in certain activities and installation. OJ L 85.

[33] Directive 94/63/EC on the control of VOC emissions from the storage and distribution of petrol (Stage I). OJ L365.

[34] Directive 2004/42/EC on the limitation of emissions of volatile organic compounds due to the use of organic solvents in certain paints and varnishes and vehicle-refinishing products and amending Directive 1999/13/EC. OJ L143.

of spraying activities, storage or petrol, to more technical measures. The Commission provides information on the very large number of measures available[35].

Given the very large number of possible measures to control VOCs, it is not possible to elaborate their individual costs here. A UK study (DEFRA, 2007) reviewed the costs for many different types of control and these vary. Which measures are most appropriate will also vary between Member States.

Costs of reducing pressures
The costs of reducing pressures are addressed in combination with other air pollutants in the following section.

5.5.5 Costs of reducing pressures

As stated in the introduction, the costs of reducing emissions for air pollutant impacts have to be considered together due to their interactions with regard to sources, impacts and generation of secondary pollutants.

Amann (2012) provides a detailed analysis of the costs of future air pollution control measures. The costs are dependent not only on measures to be taken within the EU-27, but also on measures to be taken in neighbouring non-EU countries. Additional measures in the EU-27 would involve costs of 242 million €/yr if the PRIMES Reference scenario (which achieves all the targets of the Energy and Climate package) is assumed as a starting point and non-EU countries would reduce their emissions as suggested in the CIAM 1/2011 report for the mid-case. However, if non-EU countries follow their baseline and EU countries develop along their national energy projections, additional air pollution control costs to achieve the Thematic Strategy targets would increase to 805 million €/yr. On a sectoral basis, the additional measures would involve highest costs for the agricultural sector in all cases as many other sectors are already subject to stricter control measures (Table 5-4). It is also important to note that apart from the distribution of costs between sectors, the relative implementation of current measures and relative impact of the distribution of emissions means that the costs per unit reduction of individual pollutants is unevenly distributed across the EU-27. Costs in some Member States to reduce one tonne of a pollutant are far lower than the same reduction in other Member States (Table 5-5).

The costs summarised above and presented in the tables below present a range used in the current assessment of EU and international air pollution policy development. It is not possible to provide intermediate estimates based on these results. This is because there is not a linear relationship between emissions and impacts, nor is there one between emissions and costs. The modelling is complex and it would be inappropriate to second guess results within what is a sophisticated policy analysis framework developed over many years. The modelling can assess the necessary reductions and costs for particular endpoints (eg that of the Thematic Strategy). Therefore, if other endpoints (eg a 15% or 30% reduction in critical load exceedence) should be assessed, this would need to be undertaken by use of this model and new analytical outcomes. Such endpoints would enable an examination of the reductions in pressures necessary as a precondition for achieving the Target 2

[35] http://ec.europa.eu/environment/air/pollutants/stationary/solvents/links_en.htm

restoration target. However, only full elimination of the pressure would provide the conditions necessary to ensure that the quality of all sensitive ecosystems would be maintained as also required by Target 2.

Table 5-4 Emission control costs (on top of current legislation) for achieving the targets of the Thematic Strategy on Air Pollution in 2020 (million €/yr) (Source: (Amann, 2012)

	Non-EU countries at baseline			Non-EU countries at CIAM-MID		
Sector	**PRIMES baseline**	**PRIMES Reference**	**National projections**	**PRIMES baseline**	**PRIMES Reference**	**National projections**
Power generation	14.8	9.4	93.3	8.2	3.1	42.6
Domestic	47.5	10.6	56.0	2.3	0.1	44.8
Industrial combustion	49.0	35.8	143.6	27.1	10.6	54.2
Industrial processes	24.5	26.2	86.8	8.1	2.1	29.0
Fuel extraction	0.0	0.0	0.0	0.0	0.0	0.0
Solvents	0.0	0.0	0.0	0.0	0.0	0.0
Road traffic	0.0	0.0	0.0	0.0	0.0	0.0
Off-road sources	1.9	0.0	23.2	0.0	0.0	0.8
Waste management	2.0	5.4	6.1	0.7	1.4	5.1
Agriculture	372.9	262.0	396.2	296.5	225.6	321.1
TOTAL	**512.6**	**349.3**	**805.1**	**343.0**	**242.8**	**497.6**

Table 5-5 Marginal costs of the cost-optimal set of emission reductions that meet the environmental targets of the Thematic Strategy on Air Pollution (for the PRIMES scenario), non-EU countries at baseline (€/t pollutant)

Member State	SO2	NOx	NH3	VOC
Austria	563	-	424	11
Belgium	1790	546	2514	-
Bulgaria	171	218	495	-
Cyprus	617	627	2584	220
Czech Republic	507	448	1084	11
Denmark	2013	1068	5931	-
Estonia	1126	775	1725	-
Finland	1739	887	7000	-
France	525	427	662	-
Germany	640	491	782	1
Greece	523	342	2458	1
Hungary	307	318	382	1
Ireland	617	346	1012	1
Italy	-	266	1456	11
Latvia	626	627	1235	-
Lithuania	693	748	4632	-
Luxembourg	-	753	720	11
Malta	126	-	9	-
Netherlands	640	475	1308	-
Poland	567	346	744	-
Portugal	205	140	444	1
Romania	307	276	203	-
Slovakia	-	430	495	1
Slovenia	-	401	444	11
Spain	558	244	1076	-
Sweden	5753	1068	5206	1
UK	626	459	1048	11

Conclusions

There are still major air pollution threats to Europe's ecosystems. While there have been, and will continue to be, significant declines in deposited acidity and nitrogen, critical loads will remain exceeded in some areas. Gaseous air pollutants also remain locally (ammonia) and regionally (ozone) important and can be damaging to particular protected areas. The latter is particularly difficult to control given the widespread occurrence of natural background levels.

However, current legislation and policy is delivering significant improvements and monitoring and models indicate that there will be a greater than 15% improvement in the overall area of ecosystems affected. Therefore:

- To achieve the 15% restoration target within Target 2, for acidification, no additional cost is required for air pollution control.

- For eutrophication, current legislation will not achieve the 15% target. However, it would be more than achieved under the MTRF scenario and some movement towards this is expected under the current review of air pollution policy to be undertaken in 2012-13.
- The reduction in air pollution impacts is not the equivalent of restoration – continuing damage is stopped, past damage is not necessarily restored.
- The reductions in emissions (acidification or eutrophication) do not reduce those pressures for all sensitive ecosystems. Therefore, the maintenance target within Target 2 will not be achieved through current policies or even the MTRF scenario.
- The restoration of previously damaged habitats is the subject of site-based interventions with other cost implications. This will often require either every long natural rehabilitation and/or proactive habitats management, such as removal of nutrient enriched soils (which are further described in Chapter 6).

The consequences for meeting the Target 2 objectives of maintenance and restoration need to be included within the analysis supporting the current review of air pollution policy being undertaken by the Commission. This would ensure that future air policy takes full account of these objectives and that any proposals to amend EU air policy explicitly identify if there are likely to be shortcomings in meeting the Target 2 objectives.

5.5.6 Costs and benefits of invasive alien species control and eradication measures to 2020

Based on previous Community support for controlling and eradicating already established invasive alien species [36] in the context of LIFE funding, the level of EU investment in invasive alien species management could amount to a minimum of 1.5 to 9 million €/yr[37]. The lower estimate is based on continuing earmarked funding at the same level, whilst the higher estimate is based on a funding level four times the previous contribution. Considering that the estimate is very likely to underestimate EU spending[38], and that current spending is failing to deal with the degradation to ecosystems caused by invasive alien species (EEA, 2012b), the higher estimate is considered to be most realistic in the context of achieving Target 2 by 2020. Member States could be expected to match this spending with minimum 0.75 and up to 9 million €/yr, and actual spending is likely to be higher than this. To compare: Denmark spent around €3,250,000 in one year on managing just one invasive alien species [39], without succeeding in eradicating it, a spending equivalent to 0.76 €/yr/ha over the total Danish land area (Shine et al, 2010).

EU-level measures can help to reduce costs for the EU as a whole, and the assessment shows that EU-level policy action is likely to bring more benefits (eg avoided costs) than it is estimated to cost (Shine et al, 2010), and is far less than the documented costs caused by invasive alien species currently.

[36] Those species that would be identified as 'IAS of EU concern' in the new policy framework.
[37] This is based on the EU contribution, through the LIFE programme, to managing and controlling IAS from 1992 to 2006, as per EU cofinancing arrangements, ie 50-75% match funding from EU (Scalera, 2008; Scalera, 2010).
[38] Many LIFE projects did not declare separately spending on IAS management and eradication.
[39] Local to state authority spending on control of *Heracleum mantegazzianum*, an invasive perennial species in freshwater riparian habitats, wetlands, grassland, shrub/heath etc.

6 HABITAT-SPECIFIC COSTS AND BENEFITS OF ADDRESSING PRESSURES

6.1 ARABLE ECOSYSTEMS INCLUDING TEMPORARY GRASSLANDS

Description and extent

Table 6-1 shows the area of this ecosystem type that occurs in the EU according to CLC data. The EU 2010 biodiversity baseline report includes pastures, natural grasslands, and scrub habitats within agro-ecosystems, because all these habitats require agricultural management (grazing and/or mowing) for their continued existence, but for the purpose of calculating restoration costs we analyse arable separately in this section and permanent crops, improved permanent grassland, semi-natural grassland, heathland and tundra and sclerophyllous ecosystem types in other separate sections.

The CLC class ‘arable land’ includes all areas where the dominant land use is for annual agricultural crops and other components of agricultural rotations, including non-irrigated arable land, temporary grassland (and other forage crops), fallow land, permanently irrigated land, and rice fields (Büttner et al, 2006). Arable land forms a large competent of the landscape in most EU Member States (Table 6-1), with the exceptions being Scandinavia, some western areas with high rainfall and mountainous regions. Overall, 69% of cropland is dominated by arable. The vast majority of arable land is not irrigated, but substantial areas are increasingly irrigated in southern Europe, especially in Spain. Rice production occurs in significant areas in Italy and Spain and to a lesser extent Portugal.

The CLC classes for mixed agricultural land with arable include all areas where the arable land or permanent crops or temporary grassland covers more than 50% of the area, and none of the individual crop types cover more than 75%. It typically shows complex cultivation patterns, and significant areas of natural vegetation often occur. It is important to note that a large amount of arable agriculture occurs in association with other agricultural and non-agricultural land uses, but although the total area of the CLC class for such mixed land uses is provided in Table 6-1, it is uncertain what portion of the land area is actually devoted to arable crops. Assumptions are therefore made in the calculation of the total area of arable agro-ecosystem as described with Table 6-1. **Based on our interpretation of CLC data this study assumes that the area of arable ecosystems is 121,626,223 ha.**

Table 6-1: Area of arable ecosystems (including temporary grasslands) (hectares) in the EU according to CLC data

The column shaded in grey denotes the area data that were used in this study.

Land cover class	non-irrigated arable CLC	permanently irrigated arable CLC	rice fields CLC	estimated arable crop share in mixed crop area	estimated share of arable crops in mixed farming areas	Estimated total agricultural land that comprises arable crops
CLC code	211	212	213	241	242 and 243	
Austria	1,162,612				269,262	1,431,874
Belgium	669,745				262,129	931,874
Bulgaria	3,889,138		14,319		333,717	4,237,174
Cyprus	238,685	18,903		19,348	39,735	316,671
Czech Republic	3,012,028				196,043	3,208,071
Denmark	2,785,394				130,468	2,915,862
Estonia	672,253				164,769	837,022
Finland	1,718,187				310,161	2,028,348
France	15,405,311	3,897	36,998		2,718,729	18,164,935
Germany	13,533,251				1,098,562	14,631,813
*Greece**	*1,111*	*470*	*3,710*		418	5,709
Hungary	4,932,955		11,213		136,221	5,080,389
Ireland	535,786				168,360	704,146
Italy	8,072,996	40,770	286,387	227,294	1,375,720	10,003,166
Latvia	998,536				326,227	1,324,763
Lithuania	2,221,873				459,934	2,681,807
Luxemburg	31,582			52	24,352	55,987
Malta	123				4,187	4,310
Netherlands	755,602				245,805	1,001,407
Poland	13,901,033				928,892	14,829,925
Portugal	981,182	210,724	52,836	242,629	417,187	1,904,558
Romania	8,284,985		6,839		610,492	8,902,316
Slovakia	1,676,140				106,384	1,782,524
Slovenia	112,039				156,459	268,498

Spain	9,753,705	2,199,371	144,670	84,407	2,175,600	14,357,753
Sweden	2,995,997				189,216	3,185,213
United Kingdom	6,757,319				72,791	6,830,110
TOTAL	**105,099,568**	**2,480,688**	**556,972**	**573,730**	**11,690,464**	**121,626,223**

Notes: Data relate to 2006 CLC data except where indicated as * where data are for 2000. Data for Greece are subject to significant uncertainties. The mixed crop area refers to CLC 241 mixed arable and permanent crop area, and assumes that in those Member States where areas of permanent crops are grown, there is a 60:40 ratio between arable and permanent. The mixed farming area refers to CLC 242 complex cultivation patterns and CLC 243 land principally occupied by agriculture with significant natural vegetation. The estimated share of arable crops is 40% in complex cultivation patterns CLC 242 and 25% in the agricultural mosaics CLC 243.

Arable ecosystems have a significantly impoverished biodiversity, due to their highly artificial and simplified ecology, and their high levels of frequent disturbance. Nevertheless, they still contain important biodiversity, and high numbers of some species (Stoate et al, 2009). Although many of these species are adaptable generalists and widespread they are nevertheless worthy of conservation, not least because they are commonly encountered and appreciated by people, and therefore are of high social value (Poláková et al, 2011). Furthermore, some extensively managed dry arable systems, such as those found in parts of Spain, are important habitats for globally threatened bird species such as the Great Bustard (*Otis tarda*) and Lesser Kestrel (*Falco naumanni*)(Bota et al, 2005; Pain and Pienkowski, 1997; Tucker and Evans, 1997).

The maintenance of healthy ecosystems and associated biodiversity also plays a fundamental role in supporting sustainable agricultural production. Most fundamentally all agricultural production is dependent on soils, which are not merely minerals and dead organic matter but are highly complex ecosystems with a very high level of biodiversity (Turbé et al, 2010). This soil life underpins key ecosystem services that are essential for agricultural production including decomposing plant residues and driving nutrient cycling, stabilizing soil structure, degrading pollutants, and regulating soil pests and diseases. Soils also provide important ecosystem services across the wider environment, including water retention and purification (through nutrient trapping), and carbon sequestration and storage, which has an important impact on climate regulation.

Therefore, the maintenance and restoration requirements for this ecosystem (and agriculturally improved permanent grasslands – described in section 6.3) explicitly considers pressures on these ecosystem services and the required measures to maintain and restore them, in addition to biodiversity related needs.

6.1.1 Key pressures and extent of degradation – 2010 baseline levels

Biodiversity

Changes in agricultural practices over the last 50 years or so have had well documented and widespread significant impacts on farmland biodiversity in Europe. Consequently a recent review by IEEP (Poláková et al, 2011) and others concluded that there is good evidence that the biodiversity value of agricultural land (eg the diversity and abundance of characteristic species) declines across a range of taxa groups with increasing agricultural improvement and intensification (Aebischer, 1991; Andreasen and Streibig, 2011; Billeter et al, 2008; Donald, 1998; Hendrickx et al, 2007; José-María et al, 2010; Kovács-Hostyánszki et al, 2011a; Kovács-Hostyánszki et al, 2011b; Le Féon et al, 2010; Stoate et al, 2009; Sutcliffe and Kay, 2000; van Swaay et al, 2006). Also, despite the acknowledged importance of maintaining healthy soils for agriculture, a recent analysis indicates that soil biodiversity is potentially under high pressure in nearly a quarter of the area of the EU-25 (excluding Sweden and Finland), and is under very high pressure on 8% of this area (Jeffery et al, 2010).

Although impacts vary across taxa and according to their context (eg specific habitat type and location) evidence strongly indicates that changes in the following agricultural management practices as a result of agricultural improvements, intensification and specialisation, have the most important physical and ecological impacts on arable ecosystems:

- Frequent deep ploughing and other cultivation practices that remove vegetation cover, disrupt the soil structure and lead to impacts on soil biodiversity and increased risks of soil erosion.
- Re-seeding of temporary grasslands with grass cultivars, which results in a substantial reduction in plant species diversity (and associated animal communities), increases the density and growth rates of the grassland.
- Use of fertilisers, which results in fast growing and dense crops (including temporary grasslands) and crops that provide poor breeding and feeding habitats, as well as reducing plant species diversity and associated animal communities.
- High grazing densities on temporary grasslands and other forage crops, which causes soil compaction, tramples vegetation and causes high losses of ground nesting birds.
- Cutting for silage, which results in high losses of ground nesting birds and other animals.
- Use of herbicides (and other forms of weed control), which further reduces plant species diversity (and associated animal communities).
- Use of pesticides, which reduces the abundance and diversity of invertebrate food resources for invertebrate predators (eg other invertebrates, birds and bats etc).
- Crop specialisation and reduced crop rotations, which reduces structural and ecological heterogeneity in the landscape resulting in reduced breeding and feeding options, and reduced ecological connectivity amongst habitat patches.
- Changes in timing of agricultural practices, such as crop sowing, which, for example, affects the availability of over winter food and suitable vegetation (in terms of height and density) for breeding.
- Removal of boundary habitats (such as hedgerows, stone terraces and ditches) and other non-farmed habitats (woodlots, trees and ponds), which reduces habitat diversity and connectivity amongst non-farmed habitat patches in the farmed landscape.

Other agricultural related contributory causes of farmland species declines include increased mechanisation/efficiency of cropping (eg leading to less spilt grain) and high predation rates as a result of high levels of disturbance.

It should be borne in mind that other factors that are not directly related to agriculture are also contributors to biodiversity changes in all farmland habitats, such as high predator densities, alien invasive species, hunting, disturbance, collisions with vehicles and power lines etc, external pollution sources, impacts on species in non-farmed habitats whilst on migration, or in wintering or breeding grounds, and climate change. However, evidence suggests that agricultural factors have the most significant impact on population trends in most farmland specialist species, especially those of intensively managed arable crops and improved permanent grasslands.

Ecosystem services

The most significant impacts of agriculture on the environment, other than on biodiversity, are probably on ecosystem services related to soil condition, in particular the ability of the soil to support agricultural production and its ability to store carbon and thereby mitigate climate change.

Soil degradation (ie the loss of soil structure and function), is caused by compaction, salinization, acidification, contamination, the loss of soil organic matter, and soil erosion. Soil erosion and the loss of organic matter (SOM) can threaten the long-term sustainability of farming and leads to losses of soil organic carbon (SOC) that can significantly contribute to GHG emissions and climate change (Smith et al, 2008; Smith and Olesen, 2010). The level of soil organic matter in agricultural soils influences not only soil structure but also water and nutrient holding capacity and has an important influence on soil fertility and resilience (Gobin et al, 2011).

The main causes of loss of soil organic matter include:

- conversion of grassland to arable, which results in losses of 30 tonnes of SOC stock per ha in some Member States (Zdruli et al, 2004)
- intensive tillage and deep tillage
- water and wind erosion on bare soil unprotected by a cover crop or other vegetation
- removal of crop residues and failure to replace lost organic matter, and crop monoculture or rotations without grass leys
- application of excessive amounts of inorganic nitrogen fertilizer, that results in accelerated microbial degradation of SOM
- drainage of wet carbon-rich soils
- burning of crop residues and/or wildfire; and
- degradation of landscape features including terraces, hedges, tree lines and vegetated field margins

An estimated 45% of European soils have low **organic matter content** (ie defined as having below 3.4% soil organic matter or 2% soil organic carbon), although this varies considerably between Member States (Rusco et al 2001 quoted in (Jones et al, 2004). In southern Member States, approximately 75% of soils have low organic matter content, partly reflecting the nature of the soils, the bioclimatic environmental and the extended cultivation periods in these countries (Zdruli et al, 2004). Soils in certain areas of France, the UK, and Germany also suffer from low soil organic matter content (Jones et al, 2004).

Estimates using the Revised Universal Soil Loss Equation (RUSLE) model indicate that mean rates of **soil erosion by water** in the EU-27 are 2.76 t/ha/yr, and just over 7% (115,410 km^2) of cultivated land (arable and permanent cropland) in the EU-24 (excluding Cyprus, Greece and Malta) is estimated to suffer from moderate to severe erosion (according to the OECD definition of >11 t/ha/yr) (Jones et al, 2012). The Mediterranean Member States are particularly affected. Actual soil erosion rates for tilled arable land in Europe are, on average, 3 to 40 times greater than the upper limit of tolerable soil erosion (with substantial spatio-temporal variation) (Verheijen et al, 2009a).

Furthermore, modelling of the potential risk to soil organic matter from climate change indicates that without changes to management, soil organic matter is at risk on the majority of arable soils across Europe (Gobin et al, 2011).

Estimation of the extent of pressures

An analysis of the area of agricultural land affected by key pressures on biodiversity and other ecosystem services was conducted as part of the Rural Land Management Costs study (Hart et al, 2011). To help gauge the likely extent of the main pressures resulting from agricultural intensification the study estimated the scale of intensity of agricultural management using data on the cost of inputs per hectare of utilised agricultural area (UAA). The inputs considered were purchased fertilisers and soil improvers, pesticides, other plant protection products (ie traps and baits, bird scarers, anti-hail shells, frost protection and purchased feed) (Eurostat, 2010). According to these data, the proportion of agricultural land under low intensity management equates to 41% of UAA, medium intensity 33% of UAA, and high intensity 26% of UAA. The study acknowledged that not all intensive practices associated with these indicators are necessarily detrimental to the environment, and the sensitivity and carrying capacity of different ecosystems and land uses could not be taken into account. Therefore the estimation of the area under each environmental pressure did not rely solely on these data, but also assessed pressures in relation to other evidence from more detailed studies (summarized in the Annex to the report).

It is not possible within the scope of this current study to carry out a more detailed assessment than that conducted in the Rural Land Management Costs study, and therefore its estimates of the extent of pressures that relate to arable ecosystems are used as the primary source of evidence for the estimates in Table 6-2 below. It should, however, be noted that the Rural Land Management Costs study only provided a single ‘best’ estimate of the percentage area affected by each pressure. We have therefore applied an additional +/-10% variation to provide a minimum and maximum estimate of degradation that takes into account the likely range of error in the ‘best’ estimate. It can be argued that soil salinization, soil organic matter declines and soil erosion are not in fact pressures (as listed in Table 6-2), but impacts of other activities, such as overly intensive cultivations. However, we treat them as proxies for the underlying pressures as data on the extent of the activities that lead to them are not readily available.

Table 6-2 Estimated baseline and expected 2020 percentage degradation rates, degradation levels and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

	Degradation rate 2010		Expected in 2020		
Degradation source / key pressures	**Min %/y**	**Max %/y**	**Min %/y**	**Max %/y**	**NB**
Changes in timing, specialisation & loss of rotations	0.35%	0.90%	0.35%	0.90%	1
Frequent mechanised operations	0.75%	1.60%	0.75%	1.60%	1
Use of pesticides and herbicides	0.30%	1.00%	0.30%	1.00%	1
Use of fertilisers	0.35%	0.90%	0.35%	0.90%	1
Removal of boundary / landscape features	3.50%	9.00%	0.00%	0.00%	2
Inappropriate management of boundary / landscape features	1.75%	4.50%	1.75%	4.50%	4
Irrigation and water abstraction	0.0400%	0.0600%	0.05%	0.20%	1,3
Soil salinisation	?	?	0.00%	0.00%	2
Soil organic matter decline	?	?	0.00%	0.00%	2
Soil erosion	?	?	0.00%	0.00%	2

Notes: 1 assumes a 1 - 2% annual intensification rate of remaining undegraded areas. 2 GAEC measures assumed to halt losses by 2020. 3 assumes irrigation potentially beneficial on 10% of arable land. 4 assumes 5 - 10% of remaining undegraded boundaries will be at risk

b) Degradation levels from the cumulative impacts of each key pressure

*1. Baseline degradation level minus 15%. *2. % restoration required = expected degradation level minus the maximum level to achieve target.

Degradation source / key pressures	Baseline 2010 levels		Maximum level to achieve 15% restoration target*1		Expected in 2020		% restoration required in 2020*2		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	**70.0%**	**85.0%**	**59.5%**	**72.2%**	**70.0%**	**90.0%**	**10.5%**	**17.7%**	**1**
Changes in timing, specialisation & loss of rotations	55.00%	65.00%	46.75%	55.25%	60.00%	70.00%	13.25%	14.75%	
Frequent mechanised operations	20.00%	25.00%	17.00%	21.25%	20.00%	25.00%	3.00%	3.75%	
Use of pesticides and herbicides	50.00%	70.00%	42.50%	59.50%	55.00%	75.00%	12.50%	15.50%	2
Use of fertilisers	55.00%	65.00%	46.75%	55.25%	50.00%	60.00%	3.25%	4.75%	3
Loss / inappropriate management of boundary / landscape features	55.00%	65.00%	46.75%	55.25%	55.00%	65.00%	8.25%	9.75%	4
Irrigation and water abstraction	4.00%	6.00%	3.40%	5.10%	5.00%	7.00%	1.60%	1.90%	5
Soil salinisation	2.00%	5.00%	1.70%	4.25%	2.00%	5.00%	0.30%	0.75%	
Soil organic matter decline	55.00%	65.00%	46.75%	55.25%	55.00%	65.00%	8.25%	9.75%	6
Soil erosion	60.00%	70.00%	51.00%	59.50%	60.00%	70.00%	9.00%	10.50%	6

Notes: 1, No EU wide data exist on overall degradation levels, and therefore this is based on specific pressures listed below, taking into account potential overlaps. 2, Used on virtually all non-organic arable land, but quantities and impacts vary considerably, so a wide min / max range is used. 3, Organic farms use lower rates, but many still receive high levels of manure. Use likely to decline with increasing prices. 4, Loss of boundary features expected to be addressed by GAEC requirements, but additional management measures needed. 5, Impacts are primarily on rivers and wetlands so measures are covered by the WFD (see section 6.10). 6, For this study it is assumed that anticipated GAEC measures will reduce these pressures to levels that are sufficient to maintain the ecosystem, but do not lead to restoration.

6.1.2 Key pressures and extent of degradation in 2020

The combined effects on the agricultural sector of key drivers of agricultural change (eg demand for food and energy sources, climate change and technological developments), as well as developments in the CAP and other policies, have been modelled through EU studies examining future scenarios for agriculture and the rural economy, such as Scenar 2020 (Nowicki et al, 2009) and Eururalis (Rienks, 2008). According to Poláková et al (2011), these forecast the following changes of relevance to biodiversity:

- agricultural land will decline from about 44% to 40% of EU land;
- an increase in production of food and energy crops, driven by technological change;

- declining production of livestock, especially beef, as a result of trade liberalisation;
- declining agricultural employment, especially in the new Member States;
- abandonment of marginal areas, with southern, eastern and northern countries of the EU most affected;
- declining farm incomes, especially as a result of cuts in direct payments;
- effects of Pillar 2 payments increasing productivity (farm investments), diversification, and extensification (agri-environment);
- a decline in the number of farm holdings by 25% in EU15 and 40% in EU12;
- continuing specialisation in open-field arable, horticultural and livestock-rearing/dairy systems, but at the same time a continuing role for extensive livestock-based systems with mixed cropping for fodder and fallow land; and
- continuing urbanisation in peri-urban regions.

Although these projected changes may have been affected by recent changes in drivers, the general trends seem to be robust and continuing. The EEA State of the Environment report 2010 (EEA, 2010e) notes that land-use specialisation (urbanisation, agricultural intensification and abandonment plus natural afforestation) is still a very strong trend and is expected to continue in the future. Thus, despite recent increases in world food prices, widespread abandonment of extensive marginal agriculture (ie mainly livestock systems) is still expected. Indeed, a recent JRC IPTS study using the AGLINK model quoted in the EU Resource Efficiency Roadmap Annex[40] suggests that potentially 8.6% of agricultural land will be abandoned by 2020.

However, the trends outlined above are likely to have mixed implications for arable ecosystems in the EU. The rate of biodiversity loss in improved and intensive systems in the EU-15 may decline because intensification appears to be levelling off. Although fertiliser use varies considerably, applications rates are declining in many areas as a result of price increases and better targeting of inputs. Similarly with modern crop protection approaches, pesticides tend to be used at lower rates and less frequently, for example in response to pest outbreaks rather than prophylactically. Ploughing is also being gradually replaced by reduced tillage techniques in some farming systems and areas. Nevertheless, traditional ploughing remains the norm in most conventional arable systems.

Of particular concern is the likelihood that significant further intensification of farming is likely to take place in the EU-12 Member States, especially the Baltic States, because there is considerable scope for further farm investment, restructuring and technological improvement in the region.

At the time of writing this report negotiations over the reform of the post-2013 CAP were taking place. However, it seems clear that whilst the main greening elements of the reform proposals made by the Commission (discussed in section 8.2.7) were adopted, demands by Member States and the European Parliament for increased flexibility, a reduced requirement for Ecological Focus Areas (EFAs) and a reduced Pillar 2 budget will probably result in the reforms having limited overall beneficial ecological impacts on arable

[40] SEC(2011) 1067 final http://ec.europa.eu/environment/resource_efficiency/pdf/working_paper_part2.pdf

ecosystems. Although EFAs may provide some benefits (eg if this increases the area of fallow land), this may be partly offset by declines in agri-environment funding and therefore reduced management measures in arable farmland. Although agri-environment measures can only address a small percentage of the arable area, well targeted and designed measures can have a disproportionately beneficial impact on declining species (Hart et al, 2011).

As a result of these expected changes it is anticipated that most of the pressures identified in Table 6-2 will continue, with some reduced pressures in the west being counter-balanced by intensification in the new Member States, leading to a slight overall increase in most pressures. However, modest overall declines in pressures from fertiliser use are expected as a result of their increasing costs.

It is expected that there will be increased protection of boundary and landscape features (such as hedgerows) and more effective prevention of activities that lead to soil organic matter losses and soil erosion as a result of the foreseen post-2013 CAP Pillar 1 greening measures and in particular the proposed mandatory cross-compliance requirements (see section 8.2.7). Although, at the time of writing this report, these measures have yet to be formally agreed and their impacts may depend on the final detailed legislation and its implementation, we assume for this study that they will reduce these pressures to levels that are adequate to maintain the ecosystem and its overall condition (but not by themselves to restore past losses). However, it is important to note that if the greening measures and cross-compliance are not fully incorporated into the post-2013 CAP and are inadequately implemented then the achievement of Target 2 will require additional actions to those described in this study, with significantly greater costs.

Key management and restoration measures

Within arable farming systems the broad aim is to carry out restoration measures that enhance the ecosystem with respect to key ecosystem services and the carrying capacity for farmland species, not to return it to a former low intensity ecosystem type (eg semi-natural grassland). In relation to the currently proposed restoration prioritisation framework, it equates to restoration from level 4 to level 3 (see Table 3-2). The focus of this cost assessment is also on measures that aim to mitigate the impacts of on-going arable farmland management, in particular on biodiversity and soil condition (especially concerning soil organic matter).

Several key restoration measures can reduce the loss of stored carbon in soils and increase sequestration rates. For example, a modelling exercise suggests that in a suite of basic measures suited to EU croplands, no tillage has the highest mitigation potential, followed by rotation with legumes, minimum tillage, incorporation of residues and addition of organic matter (eg compost), rotations, and vegetative cover or cover crops (Lesschen et al, 2008b). Such management measures also provide co-benefits in terms of soil biodiversity, soil fertility, workability through less compaction, water infiltration and holding capacity, and reduced erosion risk (Lal, 2004).

In addition to these key soil restoration measures, other important key measures for biodiversity include:

- reducing the use of fertilisers (which also help to reduce off-farm impacts);
- reducing the impacts of pesticides (by reducing use and/or toxicity levels and/or spectrum breadth to non-target species)
- adoption of beneficial farming practices, such as retention of stubble, incorporation of rotational fallow and 'bird-friendly' cutting practices;
- the restoration of damaged or degraded habitat features such as ponds, hedgerows and woodland for farmland species and to support broader landscape-scale conservation needs, such as reducing habitat fragmentation and facilitating climate change adaptation (Kettunen et al, 2007);
- specific measures for target species (such as the planting of field margins with seed-rich, or nectar-rich plants that provide food resources for birds and pollinators respectively).

The Rural Land Management Costs study reviewed and identified the main management options that are available under agri-environment schemes and other CAP measures that may be used to implement the broad types of mitigation measures identified above. Those that relate to arable ecosystems are summarised in Table 6-3 below, together with an indication of their potential benefits in terms of their ability to maintain and restore biodiversity and ecosystem services.

Table 6-4 summarizes the required application of each key measure to each of the key pressures on Europe's agricultural ecosystems.

Table 6-3: The range of environmental benefits provided by key management measures applied within arable ecosystems

Source: Adapted from the Rural Land Management Costs study (Hart et al, 2011, Annex 5).

Type of management required to address pressure	Biodiversity	Landscape	Water Quality	Water Quantity	Soils	Climate Change Mitigation	Climate Change Adaptation
1. Boundary / landscape feature maintenance (including of stone walls, ditches, banks, and hedges)	Y1	Y1	Y1	P	Y1	N	Y2
2. Hedgerow management & restoration (including cutting regimes, improvement of species composition, and planting).	Y1	Y1	Y2	N	N	P	Y1
3. Reduction of inputs (fertilisers and plant protection products).	Y1	N	Y1	N	Y1	P	P
4. Organic conversion and management (in accordance with certified organic standards)	Y1	N	Y1	Y1	Y1	P	N
5. Integrated Farm Management (including rational nutrient management, integrated plant protection and soil protection).	Y1	N	Y1	Y1	Y1	P	N
6. Crop rotation and diversification to reduce disease.	Y1	N	N	N	Y1	Y1	N
7. Over-winter crops / stubble management (eg maintenance/ inclusion of over-winter stubbles, catch crops and green cover crops in rotations).	Y1	N	Y1	Y1	Y1	Y1	N
8. Traditional management (including location specific traditional management practices, mosaic-like small-parcel cropping.	Y1	Y1	N	N	Y1	N	P
9. Water protection from pollution (including activities to limit nitrate leaching and soil erosion, controlled irrigation, permanent green cover).	Y1	N	Y1	N	Y1	P	Y1
10. Conversion of arable land to grassland, environmental land use change, and specification of input levels.	Y1	Y1	Y1	Y1	Y1	Y1	Y1
11. Creation of buffer strips (incl. riparian zones, buffer strips along watercourses, grass margins and field corners).	Y1	N	Y1	N	Y1	Y1	N
12. Creation of field margins for the enhancement or protection of species (incl. grassland strips of defined varieties, beetle banks).	Y1	N	Y1	N	Y1	Y1	Y1
13. Fallow (whole field).	Y1	N	Y1	Y1	Y1	Y1	N
14. Fallow (zones, eg Skylark plots).	Y1	N	N	N	N	N	N
15. Soil management (including crop rotation, reduction of soil inputs and change in ploughing regime).	Y1	N	Y1	Y1	Y1	Y1	Y2
16. Spring cultivation (to reduce nitrogen leaching).	Y1	N	Y1	N	Y1	N	N
17. Conversion of arable to grassland with specified input levels	Y1	Y1	Y1	P	Y1	Y1	P

Y1 = contributes directly to environmental objective
Y2 = contributes indirectly to environmental objective
P = has the potential to contribute to environmental objective depending on how and where it is applied.

Table 6-4: Key maintenance and restoration measures for arable ecosystems and their required percentage coverage to address key pressures

Figures in square brackets indicate alternative measures, for which the row total is considered sufficient to address the pressure. Figures in round brackets indicate measures that address more than one pressure, which are adjusted for in the costs table to avoid double-counting.

Degradation source / key pressures	GAEC[1]	Boundary maintenance	Hedgerow management	Reduction of inputs	Organic management	Organic conversion	Integrated farm management	Crop rotation	Over-winter crops / stubbles	Traditional management	Water protection	Creation of buffer strips	Creation of field margins	Fallow (whole field))	Fallow zones / patches	Soil management	Spring cultivation	Conversion to grassland
Loss / of boundary / landscape features	100																	
Inappropriate management of boundary / landscape features		4	1							1								
Changes in timing of operations / specialisation & reduced rotations					([17])	[17]	([17])	(17)	25		([17])			(7)	0.03	([50])*	10	
Frequent mechanised operations														(7)		(93)		
Use of pesticides & herbicides				([17])	([17])	[17]	([17])				([17])	2	2					
Use of fertilisers				([17])	([17])	[17]					([17])	2	2			([50])		
Salinisation											100							
Soil organic matter	100			([17])	([17])	(5)	([17])	([17])								([50])*		10
Soil erosion	100											2				([50])*		10

Note: 1, GAEC measures address maintenance requirements but do not lead to proactive restoration. * partly addressed by GAEC requirements

6.1.3 Costs of maintenance and restoration measures

The Rural Land Management Costs study (Hart et al, 2011) included the collation of information on the costs of individual agri-environment measures across approximately a third of RDPs. These data therefore provide a most detailed source of information for this study, and are used here and in the sections on agriculturally improved grasslands below. Although some agri-environment measures and their costs may have changed since the study, it is very unlikely that the changes would have a significant impact on the costs of achieving Target 2 for this ecosystem type.

It should, however, be noted that no combined measure cost estimates have been found that address all maintenance of all restoration requirements for arable ecosystems. Therefore, the costs of achieving Target 2 for this ecosystem type are entirely based on the cost of specific key measures and their requirements to address each identified key pressure. The average cost of each key measure, as calculated by the Rural Land Management Costs study, is included in Table 6-5 below.

6.1.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section provides an estimate of the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and restoring 15% of the area of the ecosystem that is expected to be degraded by each key pressure in 2020. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). Because it is not possible to ascertain overall degradation levels, or overall combined costs of maintenance and degradation, only one estimate is provided based on the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures. See section 3.4 for a detailed account of the methodology and its assumptions, and section 10.1 for a discussion of the results of the cost estimation in relation to previous studies.

The results of the estimations of costs (in addition to those expected under the reference scenario) provided in Table 6-5 indicate that the likely additional cost of **maintaining** ecological condition in arable habitats is between €220 and 556 million per year, whilst average additional cost to 2020 of achieving the **restoration target** would be between €5,960 and 6,920 million per year.

Re-creation of the ecosystem is not required to achieve Target 2 and therefore no costs for this are provided.

Table 6-5 Projected additional costs (€) in 2020 of achieving Target 2 within arable ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4. Duplicate costs are removed where "below" or" above" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 121,626,223 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020		Key measure	Sub measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max						Min	Max
ANNUAL ONGOING MAINTENANCE NEEDS IN 2020									
Loss / inappropriate management of boundary / landscape features	0.00%	0.00%	Boundary maintenance	Protection			GAEC	-	-
	1.75%	4.50%	Boundary maintenance	Management	154	4.0%	AEM	13,100,000	33,700,000
Changes in timing, specialisation & loss of rotations	0.35%	0.90%	Organic management		351	17.0%	AEM/below	25,400,000	-
	0.35%	0.90%	Integrated farm management		303	17.0%	AEM/below	21,900,000	-
	0.35%	0.90%	Crop rotation		28	17.0%	AEM	2,030,000	5,210,000
	0.35%	0.90%	Over-winter crops / stubbles		128	25.0%	AEM	13,600,000	35,000,000
	0.35%	0.90%	Water protection		324	17.0%	AEM	23,400,000	60,300,000
	0.35%	0.90%	Fallow (whole field)		152	7.0%	below	-	-
	0.35%	0.90%	Soil management		97	50.0%	below	-	-
	0.35%	0.90%	Spring cultivation		33	10.0%	AEM	1,400,000	3,610,000
Frequent mechanised operations	0.75%	1.60%	Fallow (whole field)		152	7.0%	AEM	9,710,000	20,700,000
	0.75%	1.60%	Soil management		97	93.0%	AEM	82,300,000	176,000,000

Key Pressure	% Area at risk in 2020		Key measure	Sub measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max						Min	Max
	0.30%	1.00%	Reduction of inputs		96	17.0%	below/AEM	-	19,800,000
	0.30%	1.00%	Organic management		351	17.0%	above/AEM	-	72,600,000
	0.30%	1.00%	Integrated farm management		303	17.0%	above / AEM	-	62,600,000
Use of pesticides and herbicides	0.30%	1.00%	Water protection		324	17.0%	AEM	20,100,000	67,000,000
	0.35%	0.90%	Reduction of inputs		96	17.0%	AEM/above	6,950,000	-
	0.35%	0.90%	Organic management		351	17.0%	above	-	-
Use of fertilisers	0.35%	0.90%	Water protection		324	17.0%	above	-	-
Soil salinisation	0.00%	0.00%					GAEC	-	-
Soil organic matter decline	0.00%	0.00%					GAEC	-	-
Soil erosion	0.00%	0.00%					GAEC	-	-
MAINTENANCE TOTAL								***220,000,000***	***556,000,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% Area requiring restoration in 2020		Key measure	Annual costs (€/ha/yr)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
Loss / inappropriate management of boundary / landscape features	8.25%	9.75%	Hedgerow management	503	1.0%	AEM	50,500,000	59,600,000
	8.25%	9.75%	Traditional management	401	1.0%	AEM	40,200,000	47,600,000
Changes in timing, specialisation & loss of rotations	13.25%	14.75%	Organic management	351	17.0%	AEM	962,000,000	below
	13.25%	14.75%	Integrated farm management	303	17.0%	AEM	830,000,000	below
	13.25%	14.75%	Crop rotation	28	17.0%	AEM	76,700,000	85,400,000
	13.25%	14.75%	Over-winter crops / stubbles	128	25.0%	AEM	516,000,000	574,000,000
	13.25%	14.75%	Water protection	324	17.0%	AEM	888,000,000	below
	13.25%	14.75%	Fallow (whole field)	152	7.0%	AEM	171,000,000	191,000,000
	13.25%	14.75%	Fallow (zones / pathes)	330	0.0%	AEM	1,600,000	1,780,000
	13.25%	14.75%	Soil management	97	50.0%	AEM	782,000,000	870,000,000
	13.25%	14.75%	Spring cultivation	33	10.0%	AEM	53,200,000	59,200,000
Frequent mechanised operations	3.00%	3.75%	Fallow (whole field)	152	7.0%	AEM	above	above
	3.00%	3.75%	Soil management	97	93.0%	AEM	above	above
Use of pesticides and herbicides	12.50%	15.50%	Reduction of inputs	96	17.0%	AEM	248,000,000	308,000,000
	12.50%	15.50%	Organic management	351	17.0%	AEM	above	1,120,000,000

Key Pressure	% Area requiring restoration in 2020		Key measure	Annual costs (€/ha/yr)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
	12.50%	15.50%	Integrated farm management	303	17.0%	AEM	above	971,000,000
	12.50%	15.50%	Water protection	324	17.0%	AEM	above	1,040,000,000
	12.50%	15.50%	Creation of buffer strips	454	2.0%	GAEC	-	-
	12.50%	15.50%	Creation of field margins	1009	2.0%	AEM	307,000,000	380,000,000
Use of fertilisers	3.25%	4.75%	Reduction of inputs	96	17.0%	AEM	above	above
	3.25%	4.75%	Organic management	351	17.0%	AEM	above	above
	3.25%	4.75%	Water protection	324	17.0%	AEM	above	above
	3.25%	4.75%	Creation of buffer strips	454	2.0%	GAEC	-	-
	3.25%	4.75%	Creation of field margins	1009	2.0%	AEM	above	above
Salinisation	0.30%	0.75%	Water protection	324	100.0%	AEM	above	above
Soil organic matter decline	8.25%	9.75%	Reduction of inputs	96	17.0%	AEM	above	above
	8.25%	9.75%	Organic management	351	17.0%	AEM	above	above
	8.25%	9.75%	Integrated farm management	303	17.0%	AEM	above	above
	8.25%	9.75%	Crop rotation	28	17.0%	AEM	above	above
	8.25%	9.75%	Soil management	97	50.0%	AEM	above	above
	8.25%	9.75%	Conversion to grassland	313	10.0%	AEM	314,000,000	371,000,000
Soil erosion	9.00%	10.50%	Soil management	50	50.0%	AEM	above	above

Key Pressure	% Area requiring restoration in 2020		Key measure	Annual costs (€/ha/yr)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
	9.00%	10.50%	Conversion to grassland	313	10.0%	AEM	343,000,000	400,000,000
	9.00%	10.50%	Creation of buffer strips	454	2.0%	GAEC	99,400,000	116,000,000
Key Pressure	**% Area requiring re-creation in 2020**		**Key measure**	**Annual costs costs (€/ha)**	**% area applied to**	**Instrument**	**Average annual costs over 2010-2020**	
	Min	**Max**					**Min**	**Max**
Changes in timing, specialisation & loss of rotations	13.25%	14.75%	Organic conversion	503	17.0%	AEM*1	276,000,000	below
Use of pesticides and herbicides	12.50%	15.50%	Organic conversion	503	17.0%	AEM*1	above	322,000,000
Use of fertilisers	3.25%	4.75%	Organic conversion	503	17.0%	AEM*1	above	above
Soil organic matter decline	8.25%	9.75%	Organic conversion	503	17.0%	AEM*1	above	above
							-	-
RESTORATION TOTAL							***5,960,000,000***	***6,920,000,000***

Note: 1 Organic conversion payments are made for 2 years (i.e. 503 x 2)

6.2 PERMANENT CROPS

6.2.1 Description and extent

Permanent crops include all areas dominated by vineyards, orchards of fruit trees or nut trees, hop and berry plantations, olive groves or date palm groves. The ground cover of traditionally managed permanent crops is often semi-natural grassland, although the area around the base of each tree is usually kept free of vegetation. There is a general tendency for hilly areas to have a high proportion of permanent crops which reflects the difficulties of cultivating arable crops in such areas (European Commission, 2009).

Traditionally managed old orchards, vineyards, olive groves and some other permanent crops can be of high biodiversity value due the presence of old and veteran trees (which provide important habits for invertebrates and some birds and bats), and their combinations of trees, shrubs, semi-natural grassland (which can be of high biodiversity value in itself) and bare patches, which creates mosaics of open and closed vegetation with high structural diversity. Consequently old traditionally managed permanent crops are often considered to be High Natural Value (HNV) farming systems (Baldock et al, 1993). Such systems may also provide important ecosystem services, such as the stabilisation of soils and capturing of rainfall on steep ground, as well as providing cultural and recreational benefits due to their wildlife and aesthetic qualities.

The extent of permanent crops in the EU-27 is shown in Table 6-6. Overall 25% of the CLC permanent crop area comprises vineyards, 16% is fruit or nut tree orchards or plantations, 23% is olive groves, and an additional 36% is a range of permanent crops that are part of mixed farming areas. This clearly shows that permanent crops are more abundant in Mediterranean countries; nearly 80% of the total area in the EU is in Spain, Italy, France, Portugal, and Romania. However, it should be noted that the permanent crop area in Greece is almost completely missing from the CLC statistics; although such crops are abundant there. Another study gives an area of 806,600 ha of olives in Greece (European Commission, 2010). The area of orchards may also be underestimated because Kizos et al in (Oppermann et al, 2012) estimate that orchards with scattered fruit trees cover approximately 1 million ha in temperate Europe, mainly in hilly areas. A further indication that the CLC permanent crop area is probably underestimated is that Eurostat data indicate that permanent crops in 2010 covered 10,624,330 ha[41].

Although CLC data probably underestimate the area of permanent crops in the absence of more complete and standardised data, they are used in this study for this ecosystem and therefore their area is taken to be permanent crops is **14,880,853 ha.**

[41] http://appsso.eurostat.ec.europa.eu/nui/show.do

Table 6-6: Area of permanent crops (ha) in the EU according to CLC data

The column shaded in grey denotes the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. Data for Greece are subject to significant uncertainties. The mixed crop area refers to CLC 241 mixed arable and permanent crop area, and assumes that in those Member States where areas of permanent crops are grown, there is a 60:40 ratio between arable and permanent. In the complex and mosaic areas, the estimated share of permanent crops is assumed to be 15% in the complex cultivation patterns CLC 242 and 10% in the agricultural mosaics CLC 243.

CLC land cover	Vineyards	Fruit trees	Olive groves	Estimated permanent crop share in mixed crop area	Estimated permanent crop share in complex and mosaic areas	Total permanent crop area
	221	222	223	241	242 & 243	
Austria	70,930	15			91,959	162,904
Belgium		8,176			90,778	98,954
Bulgaria	145,534	65,117			151,700	362,351
Cyprus	14,100	15,581	6,548	16,123	14,379	69,955
Czech Republic	15,894	30,625			94,372	140,891
Denmark		514			57,434	57,948
Estonia		2,036			69,043	71,079
Finland					154,985	154,985
France	1,144,825	177,016	10,584		919,507	2,251,932
Germany	129,279	121,481			385,735	636,495
*Greece**	*73*				166	239
Hungary	146,526	81,772			49,579	277,877
Ireland					73,288	73,288
Italy	528,041	400,144	1,209,285	189,412	525,923	2,890,686
Latvia		2,901			122,394	125,295
Lithuania		8,687			167,884	176,571
Luxembourg	1,464	99		44	8,658	10,273
Malta	53				2,014	2,067
Netherlands		7,288			82,210	89,498
Poland		133,724			358,094	491,818
Portugal	229,000	101,114	262,855	202,191	162,893	998,491
Romania	374,425	365,899			242,139	982,463
Slovakia	23,517	10,764			48,431	82,712
Slovenia	15,643	3,627			57,387	76,657
Spain	837,658	889,939	1,865,943	70,340	796,979	4,474,926
Sweden		1,806			85,626	87,432
United Kingdom		1,658			31,408	33,066
TOTAL	**3,676,962**	**2,429,983**	**3,355,215**	**478,109**	**4,844,963**	**14,880,853**

6.2.2 Key pressures and extent of degradation – 2010 baseline levels

Ecosystem loss

It is well known that there has been a substantial loss of traditionally managed, low input permanent crops in Europe, including very large losses of old olive groves in the Mediterranean and traditional meadow orchards in temperate Europe. These losses are being offset to some extent by an increase in permanent crops in some new areas. Nevertheless, according to Eurostat data there was a net annual loss of about 0.5% between 2005 and 2010. It is also likely that this trend is continuing, partly as a result of abandonment (as discussed below), which then triggers conversion to other land uses.

Although the loss of old growth orchards and vineyards is of concern, at least for biodiversity, landscape and other cultural reasons, the decline in the total area of permanent crops per se is probably not in most areas. It is therefore assumed that increasing the area of permanent crops by re-planting is not a high environmental priority, especially as newly planted permanent crops will be of low ecological value. Re-creation of permanent crops is therefore not considered to be necessary in significant amounts to achieve Target 2. **Pressures leading to the loss of permanent crop ecosystems and the costs of re-creation are therefore not considered in this study**. Instead the focus below is on maintaining their ecological quality and ecosystems services.

Overall degradation levels - baseline

No Habitats Directive or other EU-wide monitoring data are available on the overall status of permanent crops. Therefore we attempt to assess overall degradation levels by an analysis of the key pressures that are known to affect permanent crops, which primarily relate to agricultural intensification or abandonment processes. On the basis of such processes, permanent crops can be divided into three different states which require different approaches to the maintenance and restoration of their ecological potential (Duarte et al, 2008; Santilli et al, 2011):

- **productive sites**, which are generally on productive soils with a high crop density of small tree varieties, can be managed in a very intensive way with high use of agricultural inputs, irrigation, mechanisation etc. or can be managed using organic or integrated management methods.
- **traditional permanent crops** with low crop density mostly on marginal soils, often slopes, and interspersed or underlain by grassland, which are managed extensively for environmental, cultural and historical reasons as much as for the production value. Management is often through manual or semi-mechanised means because of access difficulties.
- permanent crops that are **abandoned or in the process of abandonment**, and in which the trees, ground cover and terraces are no longer being maintained.

Of these types, nearly all productive sites can be said to be degraded (although organic sites are generally less so) as a result of associated pressures such as the use of machinery and cultivations, which encourages soil compaction, and erosion, removal of veteran trees, irrigation and the use of fertilisers and pesticides – which are described individually below.

These pressures result in widespread degradation, especially in ecological terms and but also some ecosystems services (such as soil protection, water retention and landscape and cultural values). For contrasting reasons, as described below, abandonment also results in some ecological impacts and in some cases possible reductions in some ecosystem services (such as pollination for surrounding crops).

In conclusion it is probably reasonable to assume that all productive and most abandoned permanent crop areas are degraded. However, information on the proportion of the ecosystem that these make up is lacking. It is therefore very difficult to provide a reliable and reasonably precise estimate of overall degradation levels for the ecosystem. However, the Rural Land Use Costs Study (Hart et al, 2011) analysed a range of indictors of intensive agriculture (eg pesticide and fertiliser use) and concluded that 60% of permanent crops are under medium or high levels of management intensity. We therefore use this figure as proxy measure of the proportion of permanent crops that fall within the productive category described above. However, as this estimate is uncertain we use a broad range of possible degradation levels for this study and assume that between **50% and 70%** of the area of permanent crops was degraded in 2010.

Specific key pressures - baseline

Abandonment and lack of management

Scrub and forest regeneration on abandoned old traditional permanent cropland (and other high nature value cropland) typically results in habitat of lower biodiversity value (Suárez-Seoane et al, 2002), while abandonment also increases the risks of wildfires and soil degradation, which can have a significant impact on landscapes in the Mediterranean (Duarte et al, 2008; Lesschen et al, 2008a; Symes, 2006).

It is very difficult to quantify abandonment of permanent crops, as data on the issue are scattered and abandonment of orchards etc can be gradual and not easily detected. However, an IEEP review of evidence found that over the last few decades there has been significant but variable levels of farmland abandonment in Europe, primarily in areas where agriculture is less productive, particularly in remote and mountainous regions and areas with poor soils and harsh climates (Box 6.1). In southern Europe the annual rate of abandonment appears to have been between 0.17% and 0.8% of agricultural land.

Given their location and tendency to be on poor soils and hilly locations, it is likely that a significant proportion of the abandoned areas were permanent crops. Indeed, traditional multi-functional tree groves are considered to be at risk of being abandoned or replaced with more profitable intensive systems, for example in Portugal (Plieninger et al, 2006), Spain (Cots-Folch et al, 2006) and Greece (Allen et al, 2006). In the UK, a survey of England's traditional orchards found that 45% are in poor condition, and only 2,831 ha out of a total of 16,990 ha are included in agri-environment schemes (Burrough et al 2010 quoted in (Oppermann et al, 2012), and without support their future condition may deteriorate.

Box 6.1. Summary of recent and projected rates of agricultural abandonment in the EU

Based on *Farmland Abandonment in the EU: an Assessment of Trends and Prospects*. (Keenleyside and Tucker, 2010)

Over recent decades substantial areas of the EU have been affected by agricultural abandonment, defined here as the complete withdrawal of agricultural management such that natural succession processes are able to progress. This is largely a result of declines in the viability of extensive (low input) and small-scale agriculture systems.

Quantitative data on past abandonment are relatively sparse and mostly out-of-date. Nevertheless it is clear that that there have been significant but variable levels of farmland abandonment in Europe, primarily in areas where agriculture is less productive, particularly in remote and mountainous regions and areas with poor soils and harsh climates. For example, annual losses of Utilised Agricultural Land of 0.17% in France and 0.8% in Spain were recorded from the late 1980s to the end of the 1990s, though some of this land may have been converted to forestry rather than merely abandoned (Pointereau *et al,* 2008). In eastern Europe widespread abandonment occurred as a result of the political changes at the end of the 1980s, with abandonment estimates of 15-20% of cropland in Slovakia, Poland and Ukraine. However, some anecdotal observations suggest that significant areas of recently abandoned land have since been returned to agricultural production in the EU-12. There are, however, insufficient representative sample data to attempt an EU-wide estimation of past abandonment.

The primary drivers of abandonment are those that reduce the profitability of farming enterprises, including physical factors that limit yields and/or increase the costs of farming (eg poor soils) and economic factors such as low commodity prices and the availability of agricultural support payments. Secondary drivers may include the impacts of rural depopulation and institutional factors such as land ownership patterns and tax regimes. Although trends in the drivers of abandonment are uncertain, most are expected to remain, and some key drivers are expected to intensify, in particular as a result of increasing exposure to global agricultural markets. The economic pressures on some farming systems are also likely to be exacerbated by ongoing soil erosion and degradation over large areas of Europe as well as widespread rural depopulation.

Land abandonment is mitigated to some extent by CAP policies, including through Less Favoured Area and agri-environment payments. Cross-compliance measures also reduce the risks of complete abandonment by requiring landowners to maintain agricultural land in 'good agricultural and environmental condition'. Modelling studies suggest that despite these CAP policies there is likely to be significant levels of farmland abandonment in Europe over the next 20-30 years, particularly affecting low intensity grazing systems. However, the projected levels of abandonment vary significantly, both within and between model scenarios, ranging from 0.7% of land area by 2020 (Scenar 2020 Regionalisation Scenario; (Nowicki et al, 2009)) to 6.7% by 2030 (EURURALIS Global Cooperation Scenario).

The models are fairly consistent in their indications that the areas most at risk from abandonment will be in Finland and Sweden, the Pyrenees, north-western Spain and Portugal, the Massif Central (France), Apennines (Italy), Alps, other uplands areas of Germany and the border area of the Czech Republic and, to a lesser extent, the Carpathian Mountains.

Taking into account some of the likely weakness and constraints amongst the models Keenleyside and Tucker (2010) concluded that abandonment of 3-4% of total farmland area by 2030 is most plausible (ie of 0.1% – 0.13% per year), although the magnitude of abandonment will undoubtedly vary considerably from place to place according to local circumstances. Subsequently significant reforms of the CAP have been proposed for 2014-2020, but it seems unlikely that these will significantly change the factors causing abandonment. In fact it is expected that as a result of recent MFF decisions Pillar 2 funding for agri-environment schemes etc will decline, which may lead to an increase in abandonment rates.

However, the actual proportion of permanent crops that are abandoned every year or affected by past abandonment is not known. Therefore, in the absence of better information, it is assumed that their abandonment is in proportion to the overall rates of agricultural abandonment. Thus, assuming that abandonment rates did not change significantly between the end of the 1990s and 2010, this study assumes that in 2010 the rate of abandonment of permanent crops was between **0.17% and 0.8% per year.**

The impacts of abandonment are cumulative and will last at least 20 years in permanent crops. After that the ecosystem will have either changed to forest or at least, in most cases, be too degraded to feasibly or cost-effectively restore to a traditional system. Therefore, this study assumes that in 2010 the cumulative impacts of abandonment will affect between **3% and 16%** of the permanent crop area.

Soil erosion and degradation of terraces

Permanent crops, including many traditional vineyards and olive groves, are often grown on steep slopes. For example, in Italy, over 60% of the olive area is on sloping ground, with over 10% on slopes over 15% and 8.3% on terraces (Santilli et al, 2011). In southern and eastern Mediterranean, where plant growth is often water-limited, such terraces are tilled to create bare soil instead of vegetated ground cover, which prevents weeds competing for water and improves water penetration (Duarte et al, 2008). Various types of stone terraces and wall features are traditionally used to prevent soil erosion, but neglect results in weakened support. Stone terraces and walls that characterise many traditional permanent crop systems are culturally important and are also habitat features in themselves, for example for reptiles, birds and small mammals (Poláková et al, 2011). Maintenance of these features is labour intensive and they are at risk of being replaced with bench terracing, which involves slope levelling (Cots-Folch et al, 2006) and which can result in higher rates of soil erosion if not appropriately planned (Ramos et al, 2007).

Information on the likely extent of degraded terraces is lacking, but it is likely that the area affected by abandonment will suffer erosion for a time. The time period over which erosion may occur is uncertain, but will probably start a few years after abandonment as terraces and walls slowly degrade and then slowly cease as dense and strong perennial vegetation becomes established. Assuming this interval is about 10 years, between 40% and 70% of the permanent crop area is on sloping ground, and between 0.17% and 0.8% was abandoned annually between 2000 and 2010, then it is estimated that between 0.7% and 5.6% of the area of permanent crops in 2010 is eroded due to the lack of management of terraces.

Much more extensive erosion is likely to occur as a result of bench terracing and intensive practices, but the area impacted by these activities is not well known. For the purposes of this study we assume that because intensively managed productive permanent crops tend to have large amounts of bare soil, then between 40% and 60% of these crops on sloping are prone to erosion, which equates to 10% to 25% of permanent crops. Therefore, we assume that in total approximately **11% to 31%** of permanent crop area is affected by erosion resulting from abandonment or intensive management. As the main cause of erosion is intensification, the area of permanent crops impacted is likely to have been increasing in 2010 at a similar rate to the rate of intensification, which as described below is assumed to be **0.5% to 1.0%** per year.

Irrigation

A large proportion of permanent crops are irrigated in dry areas, especially in the Mediterranean region. For example, in Spain, currently around 28% of the olive grove area is irrigated[42]. In France, around 17% of the olive grove area is irrigated[43]. In 2003, fruit and berry orchards were 30% irrigated in Spain, 45% in Italy, 59% in France, 74% in Greece, 19% in Portugal, and only 3% in Romania; citrus was 100% irrigated in Spain, 92% in Italy, 55% in France, 98% in Greece, 80% in Portugal; vineyards were 23% irrigated in Spain, 34% in Italy, 3% in France, 36% in Greece, 8% in Portugal, 4% in Romania[44].

According to Eurostat data for 2003 (the most recent data available) there were 5,956,490 ha of irrigated permanent crops, which is about 55% of the area. However, as further described below, a recent study indicates that the area of irrigated olive groves in Spain is expected to expand from 2011 to 2020 by about 7.1%, which is about 0.8% per year (European Commission, 2012). As it is clear that irrigated areas have been expanding widely over recent years in most countries then we assume that the rate was similar in 2010 and is typical of countries with the most productive olive farming systems. However, rates may be lower in other countries and for other crops that are not so dependent on irrigation. Therefore given the uncertainty over this we round up the 0.8% figure and take it as a maximum and assume the expansion rate may be half of this, giving an assumed irrigation expansion rate for this study of **0.5% to 1.0% per year**.

According to Eurostat data for 2003 (the most recent data available) there were 5,956,490 ha of irrigated permanent crops, which is about 55% of the area. However, as noted above the area of irrigation was increasing by 0.5 – 1% per year around 2010, and therefore in this study we assume that as of 2010 between **58% and 62%** of permanent crops were irrigated.

Irrigation typically is accompanied by an overall increase in intensity of management, such as replanting with young trees and increasing the use of fertilisers and pesticides. In addition the abstraction of water from rivers and boreholes (often illegally) can have significant impacts on wetlands and groundwater levels (see section 6.10) and therefore has significant environmental impacts outside the crop area itself (European Commission, 2010).

Intensive use of fertilisers and pesticides

As with other agricultural habitats, a high use of agrochemical inputs (pesticides, herbicides nitrogen fertiliser) in permanent crops substantially lowers their value for biodiversity, as well as producing significant off-site environmental impacts (European Commission, 2010). For example, copper sulphate is used at least 5-6 times per year in intensive olive groves, and herbicides such as simazine are applied several times a year, and these compounds remain concentrated in the soil as well as running off into water courses (European Commission, 2010). Organic management of permanent crops is generally favourable for

[42] http://www.efncp.org/download/OlivefarminginAndalucia.pdf
[43] http://www.efncp.org/download/OlivefarminginFrance.pdf
[44] http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=aei_ps_ira&lang=en

biodiversity (Cardenas et al, 2006), and in 2008, at least 5% of the permanent crop area[45] in the EU was under organic management.

It is difficult to estimate extent of permanent crop areas that are affected by the intensive use of fertilisers and pesticides as there is no clear threshold beyond which impacts become significant, and these may also vary according to their context. However, it might be reasonably assumed that fertilisers and pesticides are used commonly on virtually all the productive permanent crops other those under organic management (5% of total crop area). Some use may also occur on traditional permanent crop areas, but the extent and impacts are likely to be very low. Therefore, taking into account the Rural Land Use costs study estimate (described above) that 60% of permanent crops are intensively managed and the area of organic agriculture, we assume in this study that between **50% - 60%** of the area of this ecosystem is affected by the use of fertilisers and pesticides.

The estimation of the annual rate of intensification is also problematical. However, the expansion of irrigated areas is probably a reasonable indicator of general intensification. Therefore, based on the rate of expansion of irrigation described above, we assume in this study that intensification of permanent crop management occurs at approximately **0.5% to 1.0% per year**.

Loss of veteran trees
Old trees and especially veteran trees (ie that have holes and deadwood etc) are often important habitats and refuges numerous invertebrates, as well as some bird and bat species (Davy et al, 2007). However, old fruit trees may be 10-15m high, and these are very costly- and labour-intensive to prune, and are normally not as productive as younger trees and new cultivars. Old trees are therefore often removed in favour of younger more productive trees.

Information on the impacts of this pressure is lacking, but it is likely that virtually all old trees will have been removed from productive permanent crops. Thus this pressure is likely to have impacted at least 60% of the area. In such areas it is obviously not possible to restore old mature trees, let alone veteran trees, to meet the 2020 target. However, some trees could be left to grow naturally to, and beyond, maturity in productive permanent crops to provide long-term restoration benefits. It is also likely that some traditionally managed permanent crops, which amount to about 40% of permanent crop area will also have lost some veteran trees, but the proportion is not known. For this study we assume that this could affect up to 25% of the area, giving a total area impacted across all permanent crops of some **60 – 70%.**

Currently the pressure relates mainly to traditionally managed permanent crops, and most remaining old and veteran trees within them are likely to be at risk of removal or lack of maintenance in the long-term. The annual rate of removal is likely to be at least as great as that of intensification (as described above) and could easily be much more as the removal of the trees does not require as much investment as irrigation or other forms of intensification.

[45] Ie 10% of the total organic crop area of 7,765,000 ha in 2008 as described in http://epp.eurostat.ec.europa.eu/cache/ITY_PUBLIC/5-01032010-BP/EN/5-01032010-BP-EN.PDF

Thus we assume for this study that the rate of loss of veteran trees in 2010 was between **0.5% and 2% per year**.

Loss of habitat features
Intensification of production often results in simplification of the habitat, with the loss of non-crop elements such as field margin vegetation. This is a major threat, as habitat heterogeneity underpins the rich biodiversity of traditional permanent crops, eg in vineyards in Hungary (Verhulst et al, 2004).

This pressure is likely to have already resulted in the loss of most habitat features (such as patches of scrub), other than some boundary features (such as walls and hedges) over most productive permanent crop areas (ie 60% of the permanent crop area). Some features may also have been lost in some areas of traditionally managed crops, but information on the extent of this threat is lacking. In the absence of adequate data this study assumes that **50% to 70%** of the permanent crop area was degraded in 2010 as a result of the cumulative loss of habitat features.

Information on the on-going rate of loss of habitat features in 2010 is also lacking, but we consider it is likely to be equivalent to the loss of veteran trees (for the same reasons as described above), and therefore between **0.5% and 2% per year**.

6.2.3 Key pressures and extent of degradation in 2020

Intensification to 2020

A recent study that assumes a continuing growth in the olive oil market estimates that the olive area in Spain will continue to grow by nearly 3% to 2,541,000 ha in 2020 (European Commission, 2012). Furthermore it is expected that the area of irrigated olive groves in Spain will expand from 2011 and 2020 by 90,000 ha from 28% to nearly 30% of the projected total in Spain in 2020[46] (i.e. an increase of 7.1%). This increase will be to the partial detriment of the non-irrigated area, which would decline by almost 20,000 ha over the same period. Projections for Italy suggest that the total area devoted to olive oil production will remain constant until 2020 at around 1,140,000 ha, which corresponds to the average value of the period 2000-2010, but that production will decline. For Greece, the projection is a continuation of the growth trend - although at a slower pace – from 738,000 ha in 2007 to 767,000 ha in 2020, but with a substantial decline in yield.

From the above evidence its seems highly likely that **intensification** will continue, and further conversion will occur of traditionally managed permanent crops of high importance for biodiversity and ecosystem services to intensified productive permanent crops, at least in some areas, such as Spain. However, there appears to be little information available on possible changes in the rate of intensification between 2010 and 2020. Therefore, in this study we use a broad range of estimates and, in the absence of better data, assume the rate of degradation in 2020 is as in 2010, which is **0.5% to 1.0% per year**. Thus taking

[46] Irrigated area from 681000 ha (2011) to 771000 ha (2020), non-irrigated area from 1,790,000 to 1,770,000 ha, total of 2,541,000 ha in 2020

intensification between 2010 and 2020 into account, **55-70%** of permanent crops will be degraded by the cumulative impacts of intensification in 2020.

However, it is expected that there will be increased **protection of boundary and landscape** features (such as walls) and more effective prevention of activities that lead to soil erosion as a result of the foreseen post 2013 CAP policy changes, most notably the new cross-compliance requirements and Pillar 1 greening measures (see section 8.2). Although, at the time of writing this report, these measures have yet to be formally agreed and their impacts may depend on the final detailed legislation and its implementation, we assume for this study that they will reduce these pressures to levels that are adequate to maintain the ecosystem and its overall condition. But it is important to note that if the measures are not incorporated into the post-2014 CAP and are inadequately implemented then the achievement of Target 2 will require additional actions with significantly greater costs to those estimated in this study.

Cross-compliance regulations relating to boundary and landscape features will probably not adequately protect or manage features such as patches of **scrub or tall grassy margins** etc or **veteran trees** etc. The rates of loss of these features are therefore expected to continue unchanged up to 2020. The maintenance of these features is therefore likely to need continued support through agri-environment funding in traditionally managed permanent crops.

Modelling studies suggest **agricultural abandonment** rates across Europe from 2010 to 2030 may be in the region of 0.1% - 0.13% per year (Box 6.1). But this is lower than observed for southern Europe and drivers of abandonment of old traditional orchards and olive groves etc are likely to remain or even increase. Furthermore, agri-environment funding may decline. Therefore because future abandonment trends are difficult to predict a broad range of **0.15% to 1.0% per year** is assumed for 2020.

Overall degradation in 2020

Taking these factors into account this study assumes that the overall degradation rate will increase by 5 – 10% to **60 – 80%** of the permanent crop area.

Table 6-7 provides a summary of the estimated level of overall degradation and degradation resulting from each key pressure in 2007 and expected levels in 2020.

Table 6-7 Estimated baseline and expected 2020 percentage degradation rates, degradation levels and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020		
	Min %/y	Max %/y	Min %/y	Max %/y	NB
Agricultural abandonment and under-grazing	0.17%	0.80%	0.15%	1.00%	
Soil erosion and degradation of terraces	0.50%	1.00%	0.00%	0.00%	1
Irrigation	0.50%	1.00%	0.50%	1.00%	
Intensive use of fertiliser and pesticides	0.50%	1.00%	0.50%	1.00%	
Loss of veteran trees	0.50%	2.00%	0.50%	2.00%	
Loss of habitat features - hedges and walls	0.50%	2.00%	0.00%	0.00%	1
Loss of habitat features - scrub etc	0.50%	2.00%	0.50%	2.00%	

Note: 1 In this study it is assumed that anticipated GAEC measures will reduce these pressures to levels that are sufficient to maintain the ecosystem, but do not lead to restoration.

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels		Max level for 15% restoration[1]		Expected in 2020		% restoration required in 2020[2]	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Overall	**50.0%**	**70.0%**	**42.50%**	**59.5%**	**60.0%**	**80.0%**	**17.5%**	**20.5%**
Abandonment and lack of management	3.00%	16.0%	2.55%	13.6%	4.60%	25.0%	2.05%	11.4%
Soil erosion and degradation of terraces	11.0%	31.0%	9.35%	26.3%	13.5%	36.0%	4.15%	9.65%
Irrigation	58.0%	62.0%	49.3%	52.7%	63.0%	72.0%	13.7%	19.3%
Intensive use of fertiliser and pesticides	50.0%	60.0%	42.5%	51.0%	55.0%	70.0%	12.5%	19.0%
Loss of veteran trees	60.0%	70.0%	51.0%	59.5%	62.0%	80.0%	11.0%	20.5%
Loss of habitat features - hedges and walls	50.0%	70.0%	42.5%	59.5%	52.0%	80.0%	9.50%	20.5%
Loss of habitat features - scrub etc	50.0%	70.0%	42.5%	59.5%	52.0%	80.0%	9.50%	20.5%

Notes: 1. Baseline degradation level minus 15%. 2.% restoration required = expected degradation level minus the maximum level to achieve target.

6.2.4 Key management and restoration measures

Management measures

The maintenance of the biodiversity and non-productive ecosystem service values of permanent crops is primarily dependant on the continuation of extensive traditional management practices and the avoidance of intensification or abandonment. Traditional management can involve a wide range of practices that will vary according to the crop type, location and other factors. However, those that are generally of most importance are outlined briefly below, and are included in the costing for this habitat.

Productive permanent crop trees require regular **maintenance pruning** to avoid overcrowding of branches and the development of non-productive wood**.** For example, olive trees require annual removal of water shoots (ie shoots that arise from the main tree trunk) and pruning every 2 to 3 years maximum. Permanent crops in general also require regular **replacement of lost trees** (although some veteran trees and dead trees should be preserved as habitats, see below).

Permanent crops also require **management of the ground cover.** Permanent vegetation should be maintained over the ground, except around the base of the plants, ideally of permanent semi-natural grassland; traditionally, most permanent crops (other than vines) were used both for the crop and for grazing and/or mowing. Typically under traditional extensive management about 50% of the total area of vineyards, 60% of fruit orchards, and 50% of olive groves are managed with a permanent grassy ground cover (43% of total area) whilst the remaining permanent crop area is managed for bare earth ground cover during the growing season (57% of the total area). Appropriate grazing or mowing rates are necessary for the maintenance of these HNV farming systems. In grazed permanent crops, maintenance of fencing and tree guards is needed.

Vegetation around the bases of trees can be controlled using mechanical cultivation of the ground cover in order to improve water availability and aeration of the soil for the crop tree root system. This also provides patches of bare ground as foraging habitat for birds (Schaub et al, 2010), and a certain level of disturbance is beneficial for invertebrates (Bruggisser et al, 2010). However, tilling on steep slopes results in soil erosion and ground cover should instead be maintained to prevent this (de Graaff et al, 2010; Gómez et al, 2011).

Dry stone walls and terraces should be maintained as they are more efficient than modern bench terraces at preventing soil erosion and runoff (Ramos et al, 2007), and also provide important habitat features for biodiversity, including birds, reptiles and invertebrates (Poláková et al, 2011). Other non-crop elements, such as **herbaceous or scrubby margins** or **hedges** should also be maintained in order to support biological pest control, pollinators, and biodiversity in general. The **maintenance of old veteran trees** through protection and careful pruning, where necessary, provides crucial habitat for biodiversity.

Under traditional extensive management fertilisers and pesticides are generally not used significantly. However, they are used widely and frequently in more productive systems, along with irrigation. In such systems the impacts of their use can be avoided through the adoption of **organic management** or reduced through the use of **integrated pest**

management. For example, pest control can also involve the introduction of biological control agents such as *Trichogramma* parasitoids.

Restoration measures

In accordance with the preliminary restoration framework described in section 3.2.4, it is not considered feasible or realistic to restore productive (ie intensively managed permanent crops) back to traditional systems. But instead it is assumed that measures should be taken to improve their ecological condition and provision of non-productive ecosystem services as much as possible (ie to restore 15% from degradation level 4 to level 3). This may be primarily achieved by reducing the use of fertilisers, pesticides and irrigation within productive permanent crop systems, through **organic conversion and management** or **integrated pest management**. In general, reduction of agrochemical inputs and conversion to integrated management will help improve the biodiversity value of modern intensively managed permanent crops.

Where permanent crops have been abandoned and the ground cover has become overgrown, **scrub removal / mowing / restoration grazing** may be necessary. On long-abandoned sites, machinery may be required to remove dense scrub and trees. However, in some situations browsing and grazing by large hardy livestock (such as some traditional breeds of cattle) may be sufficient to break up and remove established unwanted vegetation. The removal of scrub is also important to reduce fire risk.

Restoration of stone walls and terraces will be necessary on steep slopes where these structures have suffered damage and/or have been neglected. Erosion can also be countered with grass contour strips and drainage canals. Less productive old trees may be rejuvenated through appropriate **renovation pruning** spread over several years (pruning back hard in one year produces excessive, vigorous and unfruitful growth).

Restoration of ground cover over bare soil is also important on areas that are prone to erosion, and the re-establishment of species-rich grassland can also provide significant biodiversity benefits. The **restoration of stone walls, terraces** and other habitat features (such as non-crop native trees and ponds) will also provide benefits for some species.

Table 6-8 summarizes those that are considered to be key measures that address the key pressures on Europe’s permanent crops. In most cases the required actions are needed over the entire area that is impacted by the pressure in question (ie 100% of the degraded areas referred to in Table 6-7). However, it is likely that some measures may only need to be applied to a proportion of features, such as the rejuvenation pruning of veteran trees. In most cases the required percentage application of such actions is not well known, and the presented figures should be treated as indicative.

Table 6-8 Key maintenance and restoration and measures for permanent crops and their required percentage coverage to address key pressures

Figures in round brackets indicate measures that address more than one pressure, which are adjusted for in the costs table to avoid double-counting.

	MAINTENANCE							RESTORATION				
Degradation source / key pressures	GAEC[1]	Organic or integrated pest management	Maintenance pruning, replacement of lost trees	Management of ground cover – cultivation, grazing, mowing, fencing	Maintenance of dry stone walls and terraces	Maintenance of herbaceous/ scrubby margins, hedges etc.	Maintenance of old veteran trees	Organic conversion	Restoration of ground cover from bare soil	Scrub removal, mowing and restoration grazing	Restoration of stone walls and terraces and other habitat features	Renovation pruning
Abandonment / lack of management			100%	100%	(50%)					100%		20%
Soil erosion & terrace degradation	100%			(100%)	(50%)				100%		50%	
Intensive use of fertilisers and pesticides		100%						100%				
Irrigation		(100%)						(100%)				
Loss of veteran trees			(10%)				100%				10%	(10%)
Loss of habitat features	100%			(100%)	100%	100%					50%	

Note: 1, GAEC measures address maintenance requirements but do not lead to proactive restoration.

6.2.5 Costs of maintenance and restoration measures

Studies of the actual costs of maintaining and restoring permanent crop ecosystems seem to be lacking and therefore the assessment of costs below is almost entirely based on the cost of agri-environment measures. As discussed in section 3.4.3 there are some limitations with the use of such cost data, and they need to be interpreted carefully. In particular it should be borne in mind that agri-environment payments rates may not be sufficient to achieve high rates of uptake, especially for expensive and/or risky measures.

Combined maintenance costs

Agri-environment payments for integrated management of permanent crops amount to 425 €/ha/yr on average (ranging from 160 to 600 €/ha/yr) in Hungary, Czech Republic, Latvia, Slovakia, Slovenia, Italy, Spain and Cyprus (Department of Agriculture, 2012; Hart et al, 2011). Agri-environment schemes for extensive management pay 218 €/ha/yr on average (between 81 and 378 €/ha/yr) in Germany, Spain and Portugal. Payments of 435 €/ha/yr for permanent crops in Natura 2000 areas in Cyprus stipulate maintenance of natural vegetation field margins and light ground tillage without the use of plant protection products (Department of Agriculture, 2012). Maintenance of vineyards with traditional grape varieties and mechanical rotovation instead of herbicide application is funded at 900 €/ha/yr in Cyprus (Department of Agriculture, 2012). The preservation of vineyards defined as being on steep slopes is funded at 350 €/ha/yr in Germany (Hart et al, 2011). Agri-environment schemes for the maintenance of traditional orchards pay 94 €/ha/yr in Slovenia, 367 €/ha/yr in England and 600 €/ha/yr in Hungary (Hart et al, 2011).

Based on these costs, but excluding outliers such as vineyard maintenance in Cyprus, this study assumes an average cost of **400 €/ha/yr** for combined maintenance measures for permanent crops.

Costs of specific key maintenance measures

No agri-environment measures appear to be in place that covers the costs of **maintenance pruning** alone. This suggests that the measures are either supported through general measures or are not compensated for (because, for example, they are in fact contributing to agricultural productivity).

The costs of **managing ground vegetation cover** for erosion mitigation in Spanish olive groves average 42 €/ha/yr (from 32 to 59 €/ha) (de Graaff et al, 2010). Payments rates for such measures average 72 €/ha/yr (62-74 €/ha/yr) in Slovakia, and higher payment rates of 180 €/ha/yr are provided for soil erosion mitigation under fruit tree crops on steep slopes in Spain (Hart et al, 2011). However, the proposed post 2014 cross-compliance requirements include standards for minimum soil cover (GAEC 4) and minimum land management reflecting site specific conditions to limit erosion (GAEC 5). We therefore assume in this study that the basic maintenance of plant cover and measures to present erosion on steep slopes will be covered by cross-compliance and no longer receive significant funding under agri-environment measures.

However, it will be necessary to support environmentally beneficial management of ground vegetation, especially extensive grazing where this is feasible. The costs of appropriate grazing of orchards and other permanent crops with a permanent grassy ground cover are similar to mowing or grazing costs of semi-natural grassland. This study therefore assumes an average cost for grazing of 150 €/ha/yr and for mowing of 200 €/ha/yr (see section 6.4), and assumes that 50% of permanent crops with a permanent grassy ground cover are grazed and 50% are mown, giving an average cost of 175 €/ha/yr.

The additional costs of controlling weeds around the crop plants / trees by tillage or other mechanical means (rather than by herbicides) also needs to be considered. In olive grove this averages 59 €/ha/yr in Spain, Portugal and Greece, ranging from 41 to 77 €/ha/yr, with the highest costs incurred on steep slopes (de Graaff et al, 2010).

Thus adding and rounding the costs of grazing / mowing and tillage a total cost of the management of ground vegetation cover of 235 €/ha/yr.

Agri-environment scheme measures for the **maintenance of dry stone walls and terraces** pay 80 €/ha in Spain (Hart et al, 2011). However, the main need to maintain such features is to prevent soil erosion, and therefore it can be argued that these measures should be paid for by the landowner in order to ensure the long-term viability of their crops and to prevent soil erosion. Consequently some Member States include the maintenance of terraces as a GAEC standard. Furthermore, given the requirements under GAEC standards 4 and 5, as described above, it seems very likely that this requirement will be strengthened. Therefore it is assumed that by 2020 the maintenance of terraces would no longer be directly supported by agri-environment payments.

It is difficult to estimate the cost of **maintaining old and veteran permanent crop trees** as there does not appear to be any specific agri-environment measure for this action, as it is probably included in others or covered by GAEC standards.

Given the absence of cost data on pruning and the maintenance of veteran trees it is not possible to calculate the costs of maintaining the crop components (ie vines, trees etc) of permanent crop ecosystems by adding the costs of specific key measures. Instead we use the **combined maintenance cost of 400 €/ha/yr,** as described above.

It is also assumed the basic maintenance of boundary features (such as hedgerows and walls) and other important habitat features, will be addressed in future through cross-compliance regulations (GAEC standard 8). However, the requirement to **maintain herbaceous / scrub along the field margins** can reasonably be considered to require agri-environment support in addition to the combined cost of maintaining the cropped area. According to the Rural Land Use Costs study, the costs of maintaining such features in permanent crops is **154 €/ha/yr.**

Costs of specific restoration actions

Conversion of permanent crops to organic management can be funded through agri-environment support at **622 €/ha** on average (between 274 and 900 €/ha) in Ireland, England, France, Hungary, Belgium, Italy and Spain, while organic conversion of meadow orchards receives 100 €/ha in France (Hart et al, 2011). Organic conversion payments are normally made for two years.

The **organic management** of permanent crops averages 473 €/ha/yr (from 72 to 885 €/ha/yr) in Ireland, Germany, Belgium, Hungary, Czech Republic, Latvia, Slovenia, Italy, Spain, Portugal (Hart et al, 2011), Greece (Ministry of Rural Development and Food, 2012) and Cyprus (Department of Agriculture, 2012). However, agri-environment payment rates for integrated management, which includes controls on fertiliser and plant protection products, average about 300 €/ha/yr according to the Rural Land Use Costs study (Hart et al, 2011). The same study indicates the agri-environment payments for reduced use of fertilisers and pesticides is only about 100 €/ha/yr. However, although reductions under the latter measure may be effective in reducing pollution impacts (such as from nutrient-rich run-off) it may not meet biodiversity requirements. Therefore, in the absence of better information, for this study we take the average of the costs of organic and integrated management, which is about **390 €/ha/yr.**

The costs of **restoring permanent ground cover** on eroded soils range from **996 to 1,640 €/ha** for olive groves with different slopes and intensity of use in the same region in Spain (de Graaff et al, 2010). As described above, it is assumed that where such actions are required to alleviate soil erosion then such actions will be carried out by the farmer in order to meet cross-compliance requirements. On the other hand, the restoration may also be carried out for other purposes such as for biodiversity enhancement, in which case agri-environment funding is appropriate. This issue is dealt with in the calculation of the costs in the next section.

Where ground cover is being restored on non-eroded soils then the costs are likely to be at the lower limit of those referred to above. Therefore, for this measure the cost is taken to be about **1,000 €/ha.**

According to the Rural Land Use Costs study, the typical cost of **scrub removal in permanent crops is 41 €/ha**.

The estimation of the costs of **restoring walls and terraces** is very difficult because there are few data available on these actions and what is available is difficult to interpret. In Greece the restoration of terraces is funded at 100 €/m^3 (Ministry of Rural Development and Food, 2012). However, it is not possible to calculate the costs of restoring stone walls and terraces from these data, as the area /volume of terraces that need to be restored is unknown. As further investigations to ascertain the requirements is beyond the scope of this study we therefore refer to the costs of the constructing terraces in olive groves, which are quoted as 2,000-4,000 €/ha in Greece and Spain (de Graaff et al, 2010). As construction costs are normally higher than restoration costs, we take the wall and terrace restoration cost to be at the bottom of the construction cost range and therefore assume the typical restoration costs of **2,000 €/ha**.

The cost of **rejuvenation pruning of veteran** olive trees is estimated to be 630 €/ha in Spain, and after the pruning crop harvests will be low for about 2-3 years, but thereafter could be 30-50% higher (de Graaff et al, 2010).

Combined restoration costs

The only cost estimates for combined restoration measures that could be found related to agri-environment funded restoration of orchards in England (through tree planting, rejuvenation pruning and grazing or hay cutting). These measures cost 250 €/ha, but are unlikely to be typical of the cost of restoration of other permanent crop types, especially on eroded and hilly areas. This study does not therefore attempt to calculate restoration costs for this ecosystem on the basis of combined restoration costs.

6.2.6 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration target. There is no requirement to re-create this ecosystem to achieve Target 2, and therefore no re-creation costs. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario).

Due to inadequate cost data, no estimate is provided on the basis of the overall level of ecosystem degradation and average combined costs of maintenance. Instead, the proportion of the ecosystem that is impacted by key pressures and the cost of key measures that address each of the key pressures are used to estimate the cost of maintaining and restoring 15% of the ecosystem. See section 3.4 for a detailed account of the methodology and its assumptions, and section 10.1 for a discussion of the results of the cost estimation in relation to previous studies.

The results provided in Table 6-9 indicate that the estimated additional annual cost of **maintaining** the ecological condition of permanent crop ecosystems is between €52 and 175 million, whilst the average additional annual cost to 2020 of achieving the **restoration target** would be between €642 and 1,150 million.

Table 6-9 Projected additional costs (€) in 2020 of achieving Target 2 within permanent crop ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020 costs

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4). Duplicate costs are removed where “Included below” is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 14,880,853 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020 Min %/y	% Area at risk in 2020 Max %/y	Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020 Min	Annual costs in 2020 Max
ANNUAL ONGOING MAINTENANCE NEEDS IN 2020								
Abandonment / lack of management	0.150%	1.00%	Combined measures to maintain crops	400	100%	AEM	8,930,000	59,500,000
Soil erosion and degradation of terraces	0.000%	0.00%				GAEC	-	-
Intensive use of fertiliser and pesticides	0.500%	1.00%	Organic management or IPM	390	100%	AEM	29,000,000	58,000,000
Irrigation	0.500%	1.00%	Organic management or IPM	390	100%	above		
Loss of veteran trees	0.500%	2.00%	Combined measures to maintain crops	400	10%	AEM	2,980,000	11,900,000
Loss of habitat features - hedges and walls	0.000%	0.00%				GAEC	-	-
Loss of habitat features - scrub etc	0.500%	2.00%	Maintenance of scrubby margins etc	154	100%	AEM	11,500,000	45,800,000
MAINTENANCE TOTAL							***52,400,000***	***175,000,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% requiring restoration in 2020		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Abandonment / lack of management	2.050%	11.400%	Scrub removal and grazing	41	100%	AEM	1,250,000	6,960,000
	2.050%	11.400%	Renovation pruning	600	20%	Included below		
Soil erosion & terrace degradation	4.150%	9.650%	Restoration of ground cover	1,000	100%	AEM	61,800,000	144,000,000
	4.150%	9.650%	Restoration of stone walls	2,000	50%	AEM	61,800,000	144,000,000
Intensive use of fertilisers and pesticides	12.50%	19.00%	Organic conversion	622	100%	Included below	-	-
	12.50%	19.00%	Organic management or IPM	390	100%	Included below		
Irrigation	13.70%	19.30%	Organic conversion	622	100%	AEM	254,000,000	357,000,000
	13.70%	19.30%	Organic management or IPM	390	100%	AEM	79,500,000	112,000,000
Loss of veteran trees	11.00%	20.50%	Renovation pruning	600	10%	AEM	9,820,000	18,300,000
	11.00%	20.50%	Restoration of habitat features	2,000	10%	AEM	32,700,000	61,000,000
Loss of habitat features	9.50%	20.50%	Restoration of habitat features	2,000	50%	AEM	141,000,000	305,000,000
RESTORATION TOTAL							***642,000,000***	***1,150,000,000***

6.3 AGRICULTURALLY IMPROVED PERMANENT GRASSLANDS

6.3.1 Description and extent

For the purpose of this study, agriculturally improved grassland is defined as permanent grassland that has been so greatly modified through cultivation, reseeding, drainage, irrigation, liming, fertilisation and intensive cutting, that it has lost its semi-natural flora and fauna, and is dominated by a few grass species.

The area of this ecosystem cannot be precisely estimated because the CLC approach identifies pastures and is not able to distinguish between improved grasslands and semi-natural grasslands (described in the next section). The CLC pasture area includes all grasslands that are enclosed or parcelled, which range from semi-natural extensively used grasslands to improved grasslands. Furthermore, as indicated in Table 6-10, Member State reporting data under the CAP often differ significantly from CLC estimates of pasture area, and suggest that CLC data may underestimate the area of permanent pasture. Nevertheless, the CLC data probably provide the most consistent EU-wide dataset and which we use to ensure the comparability of cost estimates for each ecosystem type in this study. We have therefore used the CLC data for this study and have used our judgement (and reference to various HNV grassland reports and LUCAS data[47]) to estimate the proportion of pasture area in each Member State that is agriculturally improved based on knowledge about the intensity of grassland management in each area. **On this basis, this study assumes that the area of agriculturally improved permanent grasslands is 29,236,056 ha, according to the above interpretation of CLC data.**

[47] Land Use / Cover Area frame statistical Survey
http://epp.eurostat.ec.europa.eu/portal/page/portal/lucas/introduction

Table 6-10 The estimated area (hectares) of agriculturally improved grasslands, based on CLC pasture area and proportion of agriculturally improved grassland, plus the area of CAP permanent pasture for comparison

The column shaded in grey denotes the area data that were used in this study. CORINE data relate to 2006 except where indicated as * where data are for 2000. The data for Greece are subject to uncertainties. The CAP data are from the Farm Structure Survey for 2007 for BE,CZ,DK,EE,IE,LT,LU,LV,MT,NL,PO,SL and for 2005 for AT,BU,CY,DE,EL,ES,FI,FR,HU,IT,PT,RO,SE,SK,UK. The percentage of pasture area that is agriculturally improved is based on expert judgement (see annex 1 for details). It is assumed that 20% of the complex cultivation patterns CLC, and 30% of the agricultural mosaics CLC, is pasture and that the proportion of improved to semi-natural in these areas is the same as in the larger pasture area.

Land cover (ha)	CAP permanent pasture	Pastures CLC 231	Estimated % of pasture agriculturally improved	Estimated area of improved grassland in complex & mosaic areas	Estimated agriculturally improved grassland area
Austria	1,788,470	747,833	10%	16,380	91,164
Belgium	511,450	355,548	81%	132,962	420,956
Bulgaria	106,910	409,617	2%	6,877	15,070
Cyprus	440	1,171	0%	-	-
Czech Republic	909,180	700,445	20%	44,325	184,414
Denmark	201,050	56,230	90%	114,697	165,304
Estonia	273,390	247,057	50%	73,957	197,485
Finland	25,650	4,809	50%	185,919	188,323
France	8,131,420	8,695,099	85%	1,377,289	8,768,123
Germany	4,928,960	4,394,204	75%	530,777	3,826,430
Greece*	*824,250*	*391*	10%	34	73
Hungary	468,770	655,024	40%	37,712	299,722
Ireland	3,130,280	3,577,843	95%	153,299	3,552,249
Italy	3,346,950	425,884	40%	418,519	588,873
Latvia	639,520	851,391	20%	47,891	218,169
Lithuania	819,130	420,685	85%	272,123	629,705
Luxembourg	68,290	37,615	100%	16,089	53,704
Malta	0			-	-
Netherlands	820,700	1,027,260	95%	135,894	1,111,791
Poland	3,271,240	2,714,765	60%	430,575	2,059,434
Portugal	1,768,620	42,722	50%	165,005	186,366
Romania	4,530,300	2,572,829	2%	9,940	61,396
Slovakia	529,910	270,613	10%	10,989	38,050
Slovenia	288,220	116,324	10%	10,994	22,626
Spain	8,653,210	648,208	50%	762,494	1,086,598
Sweden	509,430	268,258	30%	58,058	138,536
United Kingdom	9,808,920	7,039,929	75%	51,548	5,331,495
Total	**56,354,660**	**36,281,754**		**5,064,347**	**29,236,056**

6.3.2 Key pressures and extent of degradation – 2010 baseline levels

An overview of the key pressures affecting intensively managed arable land and grasslands is provided in section 6.1.1 above, together with a discussion of collated evidence on their extent from the Rural Land Management Costs study. Based on this, and in particular a recent review of the interactions between agricultural practices and biodiversity (Polaková et al, 2011), the most important ecological impacts on agriculturally improved arable ecosystems are:

- **Conversion of permanent grassland** to arable crops and intensive temporary grassland.
- **Use of fertilisers**, which results in fast growing and dense grasslands that provide poor breeding and feeding habitats, as well as reducing plant species diversity and associated animal communities.
- **Use of herbicides**, which further reduces plant species diversity in grasslands (and associated animal communities).
- **Frequent mechanised operations** on grasslands, especially those used for silage, which often involves rolling, the spreading of fertilisers, herbicide spraying and the cutting of grasslands (often two or three times for silage), which causes soil compaction and high losses of ground nesting birds
- **High grazing densities**, which reduces plant species and structural diversity in pastures, and the abundance and diversity of associated animal communities; as well causing high losses of ground nesting birds.
- **Use of pastures and meadows for silage production**, which results in the loss of species-rich hay meadows (where practised) and reduced livestock grazing (due to longer periods in stock yards) which also reduces vegetation diversity and insects that are associated livestock and livestock dung, leading to lower food resources for birds and some other animals.
- **Removal or inappropriate / inadequate management of boundary habitats** (such as hedgerows, stone terraces and ditches) and other non-farmed habitats (woodlots, trees and ponds), which reduces habitat diversity and connectivity amongst non-farmed habitat patches in the farmed landscape.

Table 6-11 provides a summary of the key pressures leading to degradation of agriculturally improved permanent grasslands, with estimates of the extent of each pressure largely drawn from the analysis conducted as part of the Rural Land Use costs study (Hart et al, 2011).

Table 6-11 Estimated baseline and expected 2020 percentage degradation rates, degradation levels and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degredation rate 2010		Expected in 2020		NB
	Min %/y	Max %/y	Min %/y	Max %/y	
Conversion of permanent grassland	0.900%	1.900%	0.000%	0.000%	1
Use of fertilisers	0.600%	1.400%	0.500%	1.200%	2
Use of herbicides	0.700%	1.600%	0.700%	1.800%	2
Frequent mechanical operations	0.750%	1.600%	0.750%	1.800%	2
High grazing densities	0.100%	0.400%	0.100%	0.500%	2
Use of pastures and meadows for silage production	0.600%	1.400%	0.600%	1.500%	2
Removal of boundary habitats	0.350%	0.900%	0.000%	0.000%	1
Inappropriate management of boundary habitats	1.750%	4.500%	1.750%	5.000%	3

Notes: 1 GAEC & other CAP reform measures assumed to halt losses by 2020. 2 assumes a 1 - 2% annual intensification rate of remaining undegraded areas. 3 assumes 5 - 10% of remaining undegraded boundaries will be at risk

b) Degradation levels from the cumulative impacts of each key pressure

*1. Baseline degradation level minus 15%. *2. % restoration required = expected degradation level minus the maximum level to achieve target.

Degradation source / key pressures	Baseline 2010 levels		Maximum level to achieve 15% restoration target *1		Expected in 2020		% restoration required in 2020 *2		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	85.0%	95.0%	72.2%	80.7%	85.0%	95.0%	12.7%	14.2%	**1**
Use of fertilisers	30.00%	40.00%	25.50%	34.00%	25.00%	35.00%	0.00%	1.00%	
Use of herbicides	20.00%	30.00%	17.0%	25.50%	20.00%	30.00%	3.00%	4.50%	
Frequent mechanical operations	20.00%	25.00%	17.0%	21.25%	25.00%	35.00%	8.00%	13.75%	
High grazing densities	80.00%	90.00%	68.0%	76.50%	80.00%	90.00%	12.00%	13.50%	
Use of pastures and meadows for silage production	30.00%	40.00%	25.5%	34.00%	30.00%	40.00%	4.50%	6.00%	
Removal or inappropriate management of boundary habitats	55.00%	65.00%	46.7%	55.25%	55.00%	65.00%	8.25%	9.75%	2

Notes: 1, No EU wide data exist on overall degradation levels, and therefore this is based on specific pressures listed below, taking into account potential overlaps.

6.3.3 Key pressures and extent of degradation in 2020

Agriculturally improved permanent grasslands are expected to be primarily affected by the same anticipated future trends in agricultural drivers and CAP policy developments as those described for arable ecosystems in section 6.1.2. **As a result it is anticipated that most of the pressures identified above will continue, with some reduced pressures in the west being counter-balanced by intensification in the new Member States, leading to a slight overall increase in most pressures. However, modest overall declines in pressures from fertiliser use are expected as a result of their increasing costs.**

It is expected that there will be better enforced cross-compliance regulations giving protection to boundary and landscape features (such as hedgerows) and more effective prevention of the loss of permanent grasslands as a result of the foreseen post-2013 CAP Pillar 1 greening measures (see section 8.2.7). Although, at the time of writing this report, these measures have yet to be formally agreed and their impacts may depend on the final detailed legislation and its implementation, we assume for this study that they will reduce these pressures to levels that are adequate to maintain the ecosystem and its overall condition (but not by themselves to restore past losses). However, it is important to note that if the greening measures and cross-compliance are not fully incorporated into the post-2013 CAP and are inadequately implemented then the achievement of Target 2 will require additional actions to those described in this study, with significantly greater costs.

6.3.4 Key management and restoration measures

As with arable ecosystems, within agriculturally improved permanent grasslands the broad aim with respect to Target 2 is assumed to be to maintain and restore the ecosystem in terms of its carrying capacity for farmland species, and to restore non-productive ecosystem services as far as is feasible without reconverting the ecosystem. Thus the aim is not to return the ecosystem to former types such semi-natural grassland. The focus of this cost assessment is therefore on measures that aim to mitigate the impacts of on-going intensive management on these grasslands.

The Rural Land Management Costs study reviewed and identified the main management options that are available under agri-environment schemes and other CAP measures that may be used to avoid or reduce the impacts of the key pressures identified above. Those that relate to agriculturally improved grasslands are summarised in Table 6-12 below, together with an indication of their potential benefits in terms of their ability to maintain and restore biodiversity and some key ecosystem services.

Table 6-13 summarizes the required application of each key measure to each of the key pressures on Europe’s agriculturally improved grasslands.

Table 6-12: The range of environmental benefits provided by key management measures applied within arable ecosystems

Source: Adapted from the Rural Land Management Costs study (Hart et al, 2011).

Type of management required to address pressure	Biodiversity	Landscape	Water Quality	Water Quantity	Soils	Climate Change Mitigation	Climate Change Adaptation
1. Boundary maintenance (including maintenance of stone walls, ditches, banks, and hedges	Y1	Y1	Y1	P	Y1	N	Y2
2. Hedgerow management (including cutting regimes, improvement of species composition, and planting).	Y1	Y1	Y2	N	N	P	Y1
3. Grassland management (including grazing, mowing and cutting regimes, reduced fertiliser inputs).	Y1	Y1	Y1	N	Y1	P	N
4. Grazing management (including reducing and increasing grazing pressure on land).	Y1	Y1	N	N	Y1	P	N
5. Reduction of inputs (fertilisers and plant protection products).	Y1	N	Y1	N	Y1	P	P
6. Organic management (in accordance with certified organic standards).	Y1	N	Y1	Y1	Y1	P	N
7. Re-creation of grassland from arable land, environmental land use change, and specification of input levels	Y1	Y1	Y1	Y1	Y1	Y1	Y1

Y1 = Management option contributes directly to environmental objective
Y2 = Management option contributes indirectly to environmental objective
P = Management option has the potential to contribute to environmental objective depending on how and where it is applied.

Table 6-13 Key maintenance and restoration and measures for agriculturally improved grassland and their required percentage coverage to address key pressures

Figures in square brackets indicate alternative measures, for which the row total is considered sufficient to address the pressure. Figures in round brackets indicate measures that address more than one pressure, which are adjusted for in the costs table to avoid double-counting.

Degradation source / key pressures	Boundary maintenance	Hedgerow management	Grassland management	Grazing management	Reduction of inputs	Organic management	Grassland re-creation
Removal / inappropriate management of boundary habitats	1.2	0.3					
Conversion of permanent grassland to arable crops / temporary grassland							0[1]
Use of fertilisers			[(50)]		[50]		
Use of herbicides			[(45)]		[45]	[10]	
Frequent mechanical operations			(100)				
High grazing densities			[(50)]	[50]			
Use of pastures and meadows for silage production			[(50)]	[50]			

Notes: [1]. Further losses assumed to be prevented by CAP Pillar 1 greening measures, and re-creation covered under arable crops – see above.

6.3.5 Totals costs of maintenance and restoration measures

As part of the calculation of the costs of maintaining and restoring this ecosystem, the average cost of each key measure was based on the Rural Land Management Costs study data. It should, however, be noted that (as for arable ecosystems) no combined measure cost estimates were found that address all maintenance or all restoration requirements for improved permanent grassland ecosystems. Therefore, the costs of achieving Target 2 for this ecosystem type are entirely based on the cost of specific key measures and their requirements to address each identified key pressure. See section 3.4 for a detailed account of the methodology and its assumptions, and section 10.1 for a discussion of the results of the cost estimation in relation to previous studies.

The results provided in Table 6-14 indicate that the estimated annual cost (in addition to those expected under the reference scenario) of **maintaining** the ecological condition of agriculturally improved permanent grasslands is between €77 and 191 million, whilst average annual costs to 2020 of achieving the **restoration target** would be between €918 and 1,380 million.

There is no requirement to re-create this ecosystem to achieve Target 2, and therefore no re-creation costs.

Table 6-14 Projected additional costs (€) in 2020 of achieving Target 2 within agriculturally improved grassland ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). Duplicate costs are removed where "Included below" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 29,236,056 ha.

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
ANNUAL ONGOING MAINTENANCE NEEDS IN 2020								
Conversion of permanent grassland	0.000%	0.000%	Regulation	0		CAP	-	-
Use of fertilisers	0.500%	1.200%	Grassland management	232	50%	below	-	-
	0.500%	1.200%	Reduction in inputs	96	50%	below		
Use of herbicides	0.700%	1.800%	Grassland management	232	45%	below	-	-
	0.700%	1.800%	Reduction in inputs	96	45%	AEM	8,840,000	22,700,000
	0.700%	1.800%	Organic management	351	10%	below		
Frequent mechanised farming operations	0.750%	1.800%	Grassland management	232	100.0%	AEM	50,900,000	122,000,000
High grazing densities	0.100%	0.500%	Grassland management	232	50%	above	-	-
	0.100%	0.500%	Grazing management	168	50%	below		
Use of pastures and meadows for silage production	0.600%	1.500%	Grassland management	232	50%	above	-	-

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
	0.600%	1.500%	Grazing management	168	50%	AEM	14,700,000	36,800,000
Removal of boundary habitats	0.000%	0.000%	Regulation			GAEC	-	-
Inappropriate management of boundary habitats	1.750%	5.000%	Boundary maintenance	153	4%	AEM/GAEC	3,130,000	8,950,000
MAINTENANCE TOTAL							***77,600,000***	***191,000,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% Area requiring restoration in 2020		Key measure	Annual costs costs (€/ha/yr)	% area applied to	Instrum ent	Annual costs in 2020	
	Min	Max					Min	Max
Use of fertilisers	0.00%	1.00%	Grassland management	232	50%	below	-	
	0.00%	1.00%	Reduction in inputs	96	50%	below	-	
Use of herbicides	3.00%	4.50%	Grassland management	232	45%	below	-	-
	3.00%	4.50%	Reduction in inputs	96	45%	AEM	37,900,000	56,800,000
	3.00%	4.50%	Organic management	351	10%	AEM	30,800,000	46,200,000
Frequent mechanised farming operations	8.00%	13.75%	Grassland management	232	100.0%	AEM	543,000,000	933,000,000
High grazing densities	12.00%	13.50%	Grassland management	232	50%	above		
	12.00%	13.50%	Grazing management	168	50%	AEM	295,000,000	332,000,000
Use of pastures and meadows for silage production	4.50%	6.00%	Grassland management	232	50%	above		
	4.50%	6.00%	Grazing management	168	50%	above		
Removal / inappropriate management of boundary habitats	8.25%	9.75%	Hedgerow management	503	0.3%	AEM	3,640,000	4,300,000
Use of herbicides	3.00%	4.50%	Organic conversion	503	10.0%	AEM*1	8,820,000	13,200,000
RESTORATION TOTAL							***918,000,000***	***1,380,000,000***

Notes: *1 Organic conversion payments are made for 2 years (i.e. 503 x 2)

6.4 NATURAL AND SEMI-NATURAL GRASSLANDS

6.4.1 Description and extent

This section considers all natural and semi-natural grassland in the EU, which is amongst the most species-rich ecosystems in Europe and has great conservation value, but which has suffered a huge decline in past decades (EEA, 2010a). In this study these mainly comprise CLC types 'natural grassland', 'pastures' that have not been agriculturally improved (as described in the preceding section) and 'agro-forestry' (see annex 1 for more details on definitions). 'Natural grassland' includes unenclosed semi-natural or natural grassland with rocks, trees or shrubs, hedges, dry stone walls, ditches, ponds, or any other feature if the feature covers less than 25% of the area.

'Pastures' are generally enclosed by fences, hedges, ditches or dry stone walls, and also include farm structures such as livestock shelters or enclosures, small ponds, watering places or drinking troughs, as well as small patches of arable land or trees. Scrubby grassland, parklands and wooded meadows and pastures and abandoned grassland is also included if scrub covers less than 75% of the area and trees cover less than 30%. In principle, all areas of grassland with a visible field parcel structure and field boundaries are classified as pasture, and all areas of grassland without this agricultural imprint are classified as natural grassland. For example the CLC system tends to record natural grassland only in areas that are remote from settlements in upland areas whilst categorising most lowland grassland and all grassland near settlements as pasture.

Pastures vary enormously on a spectrum from very biodiversity-rich semi-natural extensively used grasslands to biodiversity-poor improved grasslands, and the CLC system does not distinguish between intensive and extensively used grassland, and cannot identify grasslands important for biodiversity and species-poor grassland. Therefore, as described in the previous section (6.3.1), for the purposes of this study judgements have been made based on available knowledge on the proportion of CLC pasture area in each Member State that is semi-natural or agriculturally improved.

Agroforestry areas are recorded in Spain, Portugal, and Italy, plus very small areas in France and Austria. In Spain, Portugal, Italy and France, the agroforestry systems are mainly grazed oak forests known as dehesa or montado (Rigueiro-Rodríguez et al, 2009). In Austria, the agroforestry area includes alpine wooded pasture. These areas are therefore considered together with the other grassland areas.

The area of natural grassland CLC, semi-natural pasture CLC and agroforestry CLC in each Member State are provided in Table 6-15, along with Annex I grasslands areas for comparison (as these will all be natural or semi-natural grasslands). There is clearly a huge variation between (and within) Member States in the proportion of permanent grassland that can be considered to be semi-natural. It is assumed that all of the CLC natural grassland and CLC agroforestry areas are semi-natural.

In addition a share of the transitional woodland-scrub area (CLC 324) is allocated to the semi-natural grassland ecosystem in order to capture abandoned grassland that is not subject to advanced scrub invasion (see Annex 1 for details).

Furthermore, in some Member States, significant areas of grassland are interspersed in the land cover classes that cover mixed agricultural landscapes ('complex cultivation patterns' or 'land principally occupied by agriculture with significant natural vegetation'). These mixed agricultural areas covered 38.7 million ha in the EU-27 in 2006 (see section 6.1), and in those Member States with predominantly mixed farming or livestock-based farming, these areas are likely to contain a significant amount of grassland. It is assumed that 20% of the complex cultivation patterns CLC, and 30% of the agricultural mosaics CLC, is pasture and that the proportion of improved to semi-natural in these areas is the same as in the larger pasture area.

Adding the areas of natural grassland, agroforestry, the share of transitional woodland-scrub considered to be semi-natural grassland, and the proportion of pasture (including pasture in mixed farming areas) that is considered to be semi-natural gives an overall total of 32.5 million ha semi-natural grassland.

It should, however, be noted that the CLC system underestimates the actual amount of permanent grassland in the EU, mainly because the aggregated land cover does not pick up small patches of grassland (smaller than around 17 ha) within other land cover types, such as forest or agricultural land. Because the CLC data are aggregated to 25 ha polygons, it fails to count these smaller patches of grassland. This is likely to include a significant proportion of the semi-natural grassland in Western Europe, where this habitat is now severely fragmented. For example, in the UK, over three-quarters of lowland semi-natural grassland sites are less than 5 ha in size and are isolated within landscapes dominated by arable, improved pasture, and/or urban sites (UK NEA, 2011a). In the mountainous and Boreal regions of Europe, semi-natural grassland patches occur in forest, heath, or rocky areas. In the Mediterranean region, semi-natural grassland occurs amongst scrub and forest.

It is possible to get some indication of which Member States contain significant amounts of grassland that is 'hidden' from the CLC data by comparing the CLC data for grasslands with the amount of permanent pasture registered under CAP. CLC registered a total of 49.3 million ha of grassland in 2006. In comparison, 56.3 million ha of permanent pasture was registered under the CAP in the same time period[48]. For example, Finland, Denmark, Slovenia, Portugal, Lithuania, Slovakia, Italy, Romania and Spain all declared significantly more grassland under CAP than is registered by the CLC data. However, the CAP figure is also an underestimate of the actual total area of grassland in the EU as in some Member States, such as Bulgaria, Cyprus, France, Hungary, Latvia and the Netherlands, large areas of semi-natural grassland are not registered under the CAP.

Despite the limitations of the CLC data, they probably provide the best available consistent EU-wide estimates of semi-natural grasslands. **Therefore, on the basis of our interpretation of CLC data, as described above, this study assumes the area of semi-natural grassland is 32,550,311 ha.**

[48] This CAP data is from Eurostat. It is from 2007 for BE,CZ,DK,EE,IE,LT,LU,LV,MT,NL,PO,SL and for 2005 for AT,BU,CY,DE,EL,ES,FI,FR,HU,IT,PT,RO,SE,SK,UK
http://epp.eurostat.ec.europa.eu/portal/page/portal/agriculture/data/main_tables.

6.4.2 Extent of biodiverse semi-natural grassland

Semi-natural grasslands span a wide range of climatic, soil and hydrological conditions, from arid to periodically or permanently waterlogged from flooding or groundwater, acidic, calcareous, saline, and oligotrophic to mesotrophic (see annex 1 for examples). The most intact (ie. floristically characteristic) semi-natural grassland communities are partly captured by the 45 Habitats Directive Annex I grassland and meadow habitats types (ETC/BD, 2008a). However there are also widespread semi-natural grassland types, such as Calthion meadows, that are not covered by the Habitats Directive. Some Member States have made efforts to improve the protection of semi-natural grassland, and have established national inventories (see annex 1 for details). However, it is not currently possible to estimate how much biodiverse semi-natural grassland there is in the EU other than by using the Habitats Directive Annex I habitat area estimates (Maes et al, 2012). This study therefore uses the total area of Annex I grassland in the EU-25 (9.3 million ha) as an estimate for the semi-natural grasslands with high biodiversity in the EU-27. This implies that 20% of Europe's permanent grassland contains a significant biodiversity value. Therefore this study assumes a total of **10 million ha of high biodiversity value semi-natural grassland** in the EU-27.

Some Member States also have large areas of low productivity rough upland grassland which have a lower biodiversity value, because they are either degraded pasture dominated by grasses of low or no forage value (*Nardus stricta*, *Molinia* spp. or *Brachypodium* spp.), or abandoned meadow land where sedges, rushes, thistles, or nettles cover more than 25% of the surface area (ETC/BD, 2008a). In this study it is assumed that this area is potentially restorable to a more ecologically valuable semi-natural grassland ecosystem (although this grassland is often the result of the degradation of heathland or the drainage of wetland).

Table 6-15 Estimated area (hectares) of natural and semi-natural grasslands, based on CLC natural grassland, agroforestry, and a proportion of pastures and mixed farming areas, and equivalent Annex I grassland habitat area

The column shaded in grey denotes the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. See annex 1 for details of ecosystem types and information on the calculation of the share of CLC type 324 that is considered to be semi-natural grassland. Natura 2000 habitat area does not include any areas in Romania and Bulgaria. It is assumed that 20% of the complex cultivation patterns CLC, and 30% of the agricultural mosaics CLC, is pasture and that the proportion of improved to semi-natural in these areas is the same as in the larger pasture area.

Land cover (ha)	Annex I grassland habitat area	Natural grassland CLC 321	Agroforestry CLC 244	Share of transitional woodland-scrub CLC 324	Pasture CLC 231	Estimated share of pasture in mixed farming	% of pasture semi-natural	Estimated share of pastures as semi-natural	Estimated natural / semi-natural grassland
Austria	760,508	598,830	60	5,360	747,833	163,804	90%	820,473	1,424,724
Belgium	16,954	886		970	355,548	164,151	19%	98,743	101,367
Bulgaria	n/a	391,599		135,972	409,617	343,867	98%	738,415	1,265,985
Cyprus	5,044	28,164		3,397	1,171	27,191	100%	28,362	59,923
Czech Republic	271,869	26,996		29,388	700,445	221,626	80%	737,657	794,041
Denmark	21,350	26,540		5,556	56,230	127,441	10%	18,367	50,463
Estonia	53,300	39,779		31,077	247,057	147,914	50%	197,485	268,342
Finland	27,210	3,572		1,405	4,809	371,838	50%	188,323	193,300
France	548,875	1,249,377	375	317,240	8,695,099	1,620,340	15%	1,547,316	2,989,466
Germany	329,524	169,025		22,731	4,394,204	707,703	25%	1,275,477	1,467,232

Land cover (ha)	Annex I grassland habitat area	Natural grassland CLC 321	Agroforestry CLC 244	Share of transitional woodland-scrub CLC 324	Pasture CLC 231	Estimated share of pasture in mixed farming	% of pasture semi-natural	Estimated share of pastures as semi-natural	Estimated natural / semi-natural grassland
Greece*	*91,288*	*993*		281	*391*	341	90%	659	1,933
Hungary	181,300	227,022		71,058	655,024	94,281	60%	449,583	747,663
Ireland	73,123	89,613		183,624	3,577,843	161,367	5%	186,960	460,198
Italy	804,100	1,464,116	175,072	188,941	425,884	1,046,298	60%	883,309	2,711,438
Latvia	14,379	5,352		131,154	851,391	239,453	80%	872,675	1,009,181
Lithuania	35,490	1,089		7,530	420,685	320,145	15%	111,124	119,743
Luxembourg	6,325			-	37,615	16,089	0%	-	-
Malta	5,800			-		4,727		4,727	4,727
Netherlands	1,982	42,114		445	1,027,260	143,047	5%	58,515	100,949
Poland	853,690	37,897		32,305	2,714,765	717,624	40%	1,372,956	1,443,158
Portugal	27,949	179,336	622,243	345,911	42,722	330,009	50%	186,366	1,333,856
Romania	n/a	318,017		164,911	2,572,829	496,990	98%	3,008,423	3,491,351
Slovakia	110,999	28,609		24,815	270,613	109,885	90%	342,448	395,872
Slovenia	256,780	20,492		4,347	116,324	109,939	90%	203,636	228,475

Land cover (ha)	Annex I grassland habitat area	Natural grassland CLC 321	Agroforestry CLC 244	Share of transitional woodland-scrub CLC 324	Pasture CLC 231	Estimated share of pasture in mixed farming	% of pasture semi-natural	Estimated share of pastures as semi-natural	Estimated natural / semi-natural grassland
Spain	4,171,569	2,642,457	2,495,438	1,269,770	648,208	1,524,988	50%	1,086,598	7,451,871
Sweden	643,912	191,577		68,441	268,258	193,527	70%	323,250	583,268
United Kingdom	132,744	1,935,999		138,621	7,039,929	68,731	25%	1,777,165	3,851,785
Total	**9,398,025**	**9,719,451**	**3,293,188**	**3,018,659**	**36,281,754**	**9,473,316**		**16,519,013**	**32,550,311**

6.4.3 Key pressures and extent of degradation – 2010 baseline levels

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures in the baseline year. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Table 6-16 provides a summary of the estimated level of habitat loss, overall degradation and the degradation resulting from each key pressure on semi-natural grasslands. The estimates are based on assessments from the Article 17 reporting (ETC/BD, 2008a) and other literature, as summarised below.

Ecosystem loss

There has been well documented losses of semi-natural grassland over much of Europe over the last 50 or more years, especially in eastern Europe (EEA, 2010a). For example, the UK lost over 90% of its lowland semi-natural grassland to intensification or conversion to arable farming (UK NEA, 2011a). The rate of loss may have slowed in recent decades for social, economic and nature protection reasons, but significant losses have continued since 2000 (EEA, 2010a).

CLC land cover change analysis[49] shows that 147,300 ha of permanent grassland were lost between 2000 and 2006 (about 0.45% of the total area of semi-natural grassland as defined and estimated in this study), mainly to urban or other artificial areas (30%), or to arable or permanent crops (32%), but also to scrub as a result of abandonment (17%) or to afforestation (8%) (EEA, 2010a).

It can be assumed that some of that area was intensive grassland, particularly near urban areas, but the greater part will have been low productivity or degraded grassland. Some semi-natural grassland areas in the lowlands are being converted to maize, oilseed rape or short-rotation coppice for biofuel production (Vandewalle et al, 2010). A map of the spread of artificial and agricultural surfaces into previously core natural or semi-natural landscapes in Europe shows that the loss of semi-natural grassland and forest to agriculture or artificial surfaces is high (3-18%) in Portugal and Spain, southern France, parts of Belgium, the Netherlands, Germany, and Hungary[50] (EEA, 2009a). In Germany as a whole, 188,000 ha (0.9%) of grassland was lost between 2005 and 2009 (Schramek et al, 2012).

In the uplands and in other marginal areas the most important current cause of loss of semi-natural grassland is long-term withdrawal of farming and abandonment, leading to colonisation by scrub and then succession to forest. The CLC land cover change analysis shows that in the last decade (2000 to 2006) forest and woodland-scrub area has expanded by 106,300 ha (EEA, 2010a), primarily as a result of the abandonment of grasslands in

[49] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

[50] NB the UK, Sweden, Finland and Greece are not assessed in this study

Southern Europe. Some abandoned grassland is being afforested, and this can alter the characteristic grassland vegetation and fauna so drastically, once the trees have grown, that the grassland habitat can be considered to be lost. However, many grassland species have a persistent seed bank, and grasslands have been successfully restored from 50 year old afforestation, provided there are some semi-natural grassland remnants nearby so that fauna and additional flora can recolonize.

Based on our interpretation of the CLC grassland loss data and the likely proportion that was semi-natural grasslands, we assume for this study an annualised loss of **0.012%** to urban or other artificial areas and **0.003%** to afforestation. The CLC data also suggest a loss of 0.013% to agricultural crops. However, the most important cause of loss of species-rich grasslands is undoubtedly agricultural improvements through for example drainage, fertilisation, ploughing and seeding with selected grasses. Such losses are not captured in the CLC land cover data because they do not distinguish between semi-natural and nor agricultural statistics on permanent grassland enable the quantification of semi-natural grassland loss to intensification. But conversion to improved grassland is much more common than to arable crops, and case study evidence from a current review of HNV grassland[51] suggests that the baseline rate of conversion to improved grasslands was probably in the order of 10 to 20 times that of arable conversion. Therefore for this study we assume an additional 0.13% to 0.26% conversion to improved grassland, giving a combined rounded annual conversion rate to arable and improved grassland in 2010 of **0.14% to 0.27%**.

Overall degradation levels – baseline

According to the Article 17 reports of Annex I habitats, 37% of the area of grassland habitats in the EU was in unfavourable condition in 2007 (ETC/BD, 2008a). If we assume that the status of the 44% of area reported with unknown status has a similar proportion of unfavourable to favourable, we can say that it is likely that 53% of the 9.63 million ha of Europe's most valuable semi-natural grassland is in a degraded state, and extrapolate this to the area of semi-natural grassland. Taking into account the high proportion of grasslands with an unknown status, this study assumes that **between 43% and 63%** of semi-natural grassland in the EU is degraded.

Specific key pressures – baseline

Scrub and weed invasion due to abandonment of extensive grazing including loss of shepherding

Many semi-natural grasslands are of marginal economic value, which together with other social factors is leading to significant agricultural abandonment especially in montane, remote and dry regions. Consequently, most semi-natural grassland degradation is due to the withdrawal or reduction of agricultural management, whether grazing or mowing. This leads to the development of tall rank grassland, invasion by perennial weedy species such as bracken (*Pteridium aquilinum*), rushes or thistles, which can be very difficult to eradicate, and later encroachment of scrub. After 10 to 15 years of abandonment it is likely that some species have disappeared from the grassland. Some extensive grasslands are also being

[51] IEEP et al, forthcoming

degraded through a mix of under- and over-grazing, caused by a decline in shepherds, who distribute livestock grazing pressure more evenly (particularly goat grazing). Abandonment of nutrient poor grassland may only lead to irreversible grassland habitat loss after a number of years or even decades, so that areas of recently abandoned grasslands can be considered to be degraded rather than lost, although nutrient-rich sites degrade more rapidly as a result of the growth of dominant grass species, scrub development, and invasive species.

In some Member States, large areas of grassland are no longer receiving CAP payments, and are therefore likely to be abandoned, or at least only partially managed. For example, in Cyprus (30%), Romania (14%), Bulgaria (11%), Estonia (16%), Finland (12%), Slovakia (9%) and Poland (6%) a significant proportion of permanent grassland that is no longer considered eligible for CAP direct support payments[52].

Grasslands where traditional grazing or hay making has been abandoned, but minimal levels of cutting are being maintained in order to maintain Good Environmental and Agricultural Condition for receipt of CAP payments, are also losing biodiversity value, because mulching is carried late in the season or even in winter, resulting in species-poor grasslands dominated by invasive weeds or alien species (King, 2010). If the mulching is carried out at the time when formerly hay was cut, the biodiversity loss will be slower (Gaisler et al, 2011).

As described in relation to permanent crops (see section) a recent review of evidence suggests that farmland abandonment in Europe over the last few decades has been in the order of 0.2% to 0.8% per year (Keenleyside and Tucker, 2010). The proportion of the abandoned farmland that is semi-natural is not certain, but abandonment mostly occurs in areas where agriculture is less productive, particularly in remote and mountainous regions and areas with poor soils and harsh climates. Semi-natural grasslands are therefore likely to be especially vulnerable. Thus, assuming that abandonment rates have not changed significantly recently study assumes that in 2010 the annual rate of abandonment was between **0.2% and 0.8%**.

Abandonment results in cumulative impacts on semi-natural grasslands, until the ecosystem changes into another type, which is typically transitional scrub and forest, which takes about 20 – 30 years. Therefore, it can also be assumed that approximately **5% to 25%** of semi-natural grassland was degraded in 2010 as a result of current and past abandonment.

Over-grazing of grasslands

In an attempt to increase agricultural production and profitability some areas of semi-natural grasslands may be subject to high stocking rates, which leads to reduced vegetation diversity and associated species (such as butterflies and other invertebrates), destruction of eggs and nests of ground-nesting birds, soil compaction (which reduces water infiltration and vegetation growth) and in severe cases erosion. These problems can exacerbated by the use of livestock feed supplements, and can be severe in areas around livestock feeders. Information on the extent of this problem is lacking, but as CAP payments are now largely

[52] Figures based on Farm Structural Survey (FSS) 2007 data. ef_lu_ovcropaa Accessed: October 2012

decoupled from livestock production, it is likely to be a relatively local problem – probably affecting **less than 5% of grasslands**.

Grassland management intensification (fertilisation and herbicide treatments)
An alternative response to falling farm profitability is intensification, rather than agricultural abandonment. The most frequent measures to improve semi-natural grasslands are the use of artificial fertilisers and occasionally herbicides. Such activities, even if only infrequent, have a profound and long-lasting effect on the vegetation and other associated biodiversity components. Indeed, even moderate use of fertilisers and occasional herbicide applications will quickly convert semi-natural grasslands into improved grasslands (which are described in the preceding chapter). However, whilst they retain a natural seedbank and key natural environmental conditions in soil, hydrology etc., it may be possible to restore the semi-natural vegetation with suitable measures.

Information on the extent of semi-natural grassland that are degraded, but not converted, as result of agricultural intensification, is not readily available as changes are subtle, complex and context-specific. However, intensification seems to be uncommon due to constraining factors, such as inadequate funds for investment and the poor soils, harsh climates and difficult terrain associated with many semi-natural grasslands. Furthermore, there may be environmental considerations, such as within Natura sites, that prevents agricultural improvements. Therefore, for the purpose of this study, it is assumed that baseline intensification rates were about **1% to 2% per year**, with **10% to 20%** impacted as a result of cumulative intensification that has degraded the ecosystem (but not converted it to improved grassland).

Hydrological modification and drainage
Humid and wet semi-natural grassland is degraded due to changes in hydrological conditions (drainage, separation from natural flooding or changes in groundwater characteristics) and river engineering that cuts off grassland from river floods, and prevents periodic rises in the groundwater table. Fen meadows that are permanently waterlogged below the surface are highly sensitive to the deterioration of groundwater quality and quantity. Under the Habitats Directive, Member States in the EU-25 reported 743,195 ha of humid and wet grassland habitat types (8% of the Annex I grassland area), and 70% this area was reported as being in unfavourable conservation status with the rest as unknown.

In this study it is therefore assumed that **between 5% and 6%** of the semi-natural grassland area is degraded due to hydrological modifications. Baseline rates of further hydrological modification are uncertain, but likely to be low due to the low profitability of livestock farming on semi-natural grasslands (though hydrological modification as part of conversion to improved grassland being probably more common). In the absence of better data this study assumes a baseline hydrological modification rates of **0.1% to 1% per year.**

Alien invasive species
Invasive alien species are not generally a problem on grasslands relative to other habitats, but some abandoned grassland types are being invaded by fast-growing herbs such as *Bunias orientalis* or shrubs such as *Rosa rugosa* or tree species such as *Ailanthus altissima* or *Robinia pseudoacacia*. The EU-wide extent and significance of the impacts of such invasive

species is however not well documented, and it is not possible within the scope of this study to quantify them.

Wildfire damage
Some grassland areas, especially in Southern Europe, are prone to fires. These can be ignited by lighting, but are more often the result of human activities, most of which are accidental. Fires may also be started deliberately on grasslands to clear scrub, weeds etc. (Martínez et al, 2009). Many grassland fires will not have significant impacts of grasslands as fires are in fact a natural component of such ecosystems (Bond and Keeley, 2005; Keeley et al, 2012; Wright and Bailey, 1982). However, large and intense fires can result in the total loss of surface vegetation, and even roots and organic matter in the upper layers of the soil (Tucker, 2003). This can lead to erosion and irreversible impacts on the ecosystem. Large and intense fires tend to develop where tall grass and litter builds up and scrub becomes established. Therefore the maintenance of grassland and extensive livestock grazing can play an important role in reducing the incidence of large and intense wild fires and their impacts.

Information on the general extent and frequency of fires is available, and suggests that the incidence of large wildfires is increasing (Moreira et al, 2011), but the percentage of semi-natural grasslands that are detrimentally impacted is highly uncertain. It is therefore not possible to quantify the impact of wildfire damage on the degradation of grasslands.

Loss of associated habitat features
Much of the biodiversity of semi-natural grasslands is associated with the mosaic of other features found in and around the grasslands, such as rocks, trees or shrubs, hedges, dry stone walls, ditches, or ponds. The loss or degradation of these features, for example through the consolidation of small holdings in Romania and Bulgaria and the loss of boundary habitats, significantly reduces the value of the grassland for many species.

Information on the extent of this pressure is lacking but it is likely to be widespread and therefore it would not be appropriate to discount it in this study. Therefore, we assume a wide range of possible impacts from this pressure, and take it to be between **20% and 60%** of the habitat area. Baseline rates of loss are also uncertain, but are known to have been significant (despite existing GAEC requirements to protect them) and for this study we assume they were between **1% to 3% per year.**

Eutrophication from air pollution
It is important to highlight that one of the most important pressures on semi-natural grasslands is deposition of atmospheric nitrogen which leads to eutrophication and significant vegetation changes. However, this pressure is best dealt with through wide-scale at source measures, which are described in section 5.5.

6.4.4 Key pressures and extent of degradation in 2020

Ecosystem loss from 2010 to 2020

Although the CAP and the EIA framework both contain elements that are designed to protect permanent grassland, in practice the policy mechanisms have enough loopholes that they do not prevent the intensification of grassland or the conversion of grassland into agriculture or forestry, so long as some other areas of grassland are being created elsewhere, without reference to its biodiversity value (Beaufoy et al, 2011). However, it is expected that improved enforcement of existing measures will occur and the European Commission's CAP reform proposals will provide stronger protection for permanent grasslands. Quantification of the impacts of these changes is not easy, but we assume that rates of loss over the 2010-2020 period will be half those of 2000-2010.

Specific pressures in 2020

Abandonment trends

A high proportion of the remaining semi-grassland areas are located in mountains, and rely for their survival on the continuance of extensive farming practices that have virtually disappeared in the lowlands. The challenge is how to avoid large-scale abandonment in the context of existing production systems. It is foreseen that extensive grazing will continue to decrease in the future, driven by socio-economic and demographic changes in rural populations, and the declining economic viability of extensive meat and wool production (Poláková et al, 2011), but it is difficult to quantify the likely changes in pressures resulting from agricultural activities as much will depend on market forces and the currently debated reforms of the CAP.

Modelling studies have attempted to quantify likely changes in land use trends in the EU but they may no longer be valid as result of the changes in commodity prices and other social and economic drivers of land use change in recent years and possible changes in the CAP. As discussed in Chapter 8 the outcomes of the current debates on reform of the CAP are uncertain, but the proposed Pillar 1 greening measures may have a significant impact on permanent pasture. However, substantial detrimental biodiversity impacts would occur on semi-natural grassland ecosystems, as a result of increased abandonment and reduced conservation management, if agri-environment and Natura funding under Pillar 2 declines. Improved targeting and guidance could increase the efficiency and effectiveness of agri-environment measures etc, but this could be offset by further increasing social-economic pressures and climate change impacts on the viability of extensive agriculture in marginally economic ecosystems. A significant increase in Pillar 2 funding would be required to reduce these existing impacts to maintain semi-natural grassland ecosystems; and this seems highly unlikely in the current EU economic climate.

In the absence of better information, this study assumes that abandonment and under-grazing impacts will continue but their rate will change little. However, some increase may occur as economic pressures are likely to increase and policy measures may not fully compensate for these. Abandonment rates are therefore taken to be between **0.2% and 1.0% per year** with approximately **5% to 30%** of semi-natural grassland degraded as a result of current and past abandonment.

Other pressures

Over-grazing is likely to decline to negligible levels due to economic factors and better enforcement of cross compliance regulations. Impacts from the intensification of grassland management, fires and the loss of habitat features are expected to decline, primarily as a result of better protection measures, guidance on environmental management, improved implementation of CAP cross-compliance standards, and environmental and economic constraints on farming profitability – however, these pressures will remain to some extent.

It is also considered likely that drainage activities will decline by 2020 due to better protection measures, guidance on environmental management and constraints on farming profitability, which make such capital investments increasingly uneconomic. However, this is likely to be offset by increasing impacts of climate change, which are already resulting in observable impacts on wet grassland. It is not possible to quantify the net impact of these expected opposing changes in drainage and climate change and therefore in the absence of better information this study assumes that these pressures will remain at baseline levels in 2020.

Overall degradation levels to 2020

Taking all the specific pressure trends into account, and respective foreseen measures to address them, it seems possible that overall degradation levels may fall slightly, but it is more likely that a small increase will occur primarily as a result of abandonment (which could be larger if significant cuts in agri-environment funding result from the current CAP reforms). Therefore, this study assumes that as a reference scenario overall levels of degradation of semi-natural grasslands in 2020 will be between **40% and 65%**.

Table 6-16: Estimated baseline and expected 2020 percentage degradation rates, degradation levels and corresponding requirements to achieve a 15% restoration target and conversion rates

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020		NB
	Min %/y	Max %/y	Min %/y	Max %/y	
Overall	**3.00%**	**12.0%**	**1.00%**	**6.00%**	
Abandonment and under management	0.200%	0.800%	0.200%	1.000%	
Over-grazing	1.000%	5.000%	0.000%	0.000%	1
Grassland management intensification	1.000%	2.000%	0.500%	1.800%	
Hydrological modification	0.100%	1.000%	0.100%	1.000%	
Inappropriate burning and wildfires	?	?	?	?	
Loss of habitat features	1.000%	3.000%	0.100%	2.000%	

Notes: 1, it is assumed that anticipated GAEC measures will reduce these pressures to acceptable levels.

b) Degradation levels from the cumulative impacts of each key pressure

*1. Baseline degradation level minus 15%. *2. % restoration required = expected degradation level minus the maximum level to achieve target.

Degradation source / key pressures	Baseline 2010 levels		Maximum level to achieve 15% restoration target*		Expected in 2020		% restoration required in 2020		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	**43.0%**	**63.0%**	**36.5%**	**53.5%**	**40.0%**	**65.0%**	**0.00%**	**0.00%**	
Abandonment and under management	5.00%	25.0%	4.25%	21.2%	5.00%	30.0%	0.75%	8.75%	
Over-grazing	1.00%	5.00%	0.850%	4.25%	0.00%	0.00%	0.00%	0.00%	1
Grassland management intensification	10.0%	20.0%	8.50%	17.0%	5.00%	10.0%	0.00%	0.00%	2
Hydrological modification	5.00%	6.00%	4.25%	5.10%	5.00%	6.00%	0.75%	0.90%	
Inappropriate burning and wildfires			0.000%	0.00%			0.00%	0.00%	
Loss of habitat features	20.0%	60.0%	17.0%	51.0%	20.0%	60.0%	3.00%	9.00%	2
Invasive species			0.000%	0.00%			0.00%	0.00%	

Notes: *Baseline degradation level minus 15%. 1. In this study it is assumed that anticipated GAEC measures will reduce these pressures to acceptable levels and pro-active restoration measures are not necessary. 2. The rate of intensification, but the overall area impacted in 2020 will not change significantly due to the need for proactive measures to reverse cumulative impacts.

c) Conversion rates

Conversion rates	Conversion per year 2000-2010		Conversion per year 2010-2020		Total converted over 2000-2020		% re-creation required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Conversion to urban and economic sites	0.010%	0.015%	0.005%	0.008%	0.150%	0.225%	0.023%	0.034%
Conversion to forestry	0.001%	0.005%	0.000%	0.003%	0.010%	0.075%	0.002%	0.011%
Conversion to crops & improved grassland	0.140%	0.270%	0.140%	0.270%	2.80%	5.40%	0.420%	0.810%
Total	**0.151%**	**0.290%**	**0.145%**	**0.280%**	**2.96%**	**5.70%**	**0.444%**	**0.855%**

6.4.5 Key management and restoration measures

A basic distinction can be made between the large areas of semi-natural grassland that remain in uplands and mountains, which require support for existing extensive production systems, and the drastically reduced and fragmented remnants of semi-natural grassland in the lowlands, which are often no longer part of effective production systems and require specific nature conservation measures including strong protection, publicly funded conservation management, and sometimes land purchase (the cost of which is not assessed in this study).

However, whether upland, lowland, extensive or fragmented, management of semi-natural grassland almost always involves extensive grazing, often in combination with traditional mowing to produce hay crops [ref]. But it is important to note that the ecological value of semi-natural grassland often depends on particular practices, such as the period of grazing, stock type and timing of cutting. These practices need to be adjusted to the conservation needs of the specific area of grassland, which will vary according to its type, condition and other factors. Restoration and re-creation measures need to provide the environmental conditions typical of that grassland type, especially low fertility soils and appropriate hydrological regimes (dry, wet, periodically flooded etc), and interventions, where needed, to create disturbance to allow propagation of desirable grassland species.

Grazing needs to be carried out at the correct stocking rate for the grassland type – stocking rates may need to be adjusted as habitat changes, eg an initial period of more intensive grazing will suppress dominant vegetation. The correct grazing period also varies by grassland type – eg summer or winter grazing only. In some areas traditional transhumance grazing or droving should be maintained. Rotational grazing may be appropriate on some grassland types, but requires more intensive labour for movement of livestock and fencing. The use of hardy native breeds that can graze low quality forage and remain outside during winter is a useful measure; extensive all-year-grazing with large livestock is most effective in reducing over-standing biomass (litter) and dominant stands of undesired species in many grassland types and produces near-natural vegetation mosaics. Use livestock type or mix of livestock that creates desired vegetation structure. Sheep, goats, equines, cattle, geese, reindeer (perhaps in combination with wild animal grazing eg rabbits, deer, ibex or chamois) all have different effects on the vegetation. Shepherding is a key measure on open grassland. Importantly, grazing requires the installation and maintenance of fencing and gates, cattle grids, water supply for livestock, livestock shelters or enclosures. However, livestock corralling, especially sheep folding, is inappropriate on many types of grassland.

In addition to grazing, the management of semi-natural grassland often also requires **mowing / cutting** as part of the production of hay crops. It is important to ensure that the cutting is carried out at appropriate times, such as to birds and invertebrates to breed, and desired plants to flower and set seed. Conversely mowing (or grazing) at certain times can be used to control undesirable species. Mosaic mowing or staggered mowing leaving, for example, 5-10% of an area uncut until the following year (and a different area left each year), and/or retention of unmown strips or patches, is important for maintaining meta-populations of invertebrate species and other species that benefit from refuges in the uncut grass. Cuttings should be removed from the site to prevent nutrient enrichment and suppression of vegetation regrowth, but it is also beneficial to let the hay lie to disperse seeds and invertebrates. Mowing machinery needs to be appropriate for habitat, for example to avoid soil compaction on wet grasslands, and to minimise killing of birds and animals in mowing blades, for example using a horizontal bar mower set high. Some meadows, such as mountain areas, need to be cut manually using a brush cutter or a scythe. It is often appropriate to follow cutting with a period of grazing, depending on the grassland type.

Fertiliser and manure application should only be carried out in certain grassland types where it does not have a negative effect. In certain places hay meadows can tolerate occasional low level manure applications to maintain their characteristic species composition, but in most European regions it is not recommended to fertilise hay meadows at all.

Grassland maintenance also requires the periodic **control of weeds and invasive species**. Temporary heavy grazing can be used to reduce perennial weeds eg to trample bracken[53], or in spots with heavy infestation it is recommended to cut or top weeds before they flower (but if possible not during the bird nesting season). It may be necessary to chain harrow or roll weeds (but again not during the bird nesting season).

Wild fire damage on grasslands can be managed through **fire prevention and management measures**, such as the installation and maintenance of fire breaks, and in fact grasslands are themselves important fire breaks in between forest and scrub. Therefore the primary measure to avoid intensive fires is the maintenance of grazing and cutting regimes, to avoid the encroachment of scrub and the build-up of dry leaf litter. Where this is not possible, controlled burns may be beneficial for some grassland types, but these need to be carried out carefully and only in appropriate locations and conditions.

In addition measures can be taken to prevent the outbreak of accidental and intentional fires, such as through the provision of warning signs (indicating penalties for arson), fire patrols, awareness raising in local communities, and investigation of deliberately started fires and prosecution of arsonists. The provision of basic fire control equipment (eg beaters) can also help to avoid small accidental fires becoming larger uncontrollable fires.

The **maintenance of hydrological characteristics** is important for wet grassland, including water level control and the maintenance of ditches, sluices, dams etc.

The **retention and management of farmland features** associated with grasslands includes a pruning regime for hedges, maintenance of dry stone walls or other valuable farmland features, and maintenance of ponds and ditches.

Restoration of degraded grassland

Restoration of degraded species rich grassland requires some or all of the following measures, as well as the management measures listed above (especially grazing).

[53] http://www.liv.ac.uk/researchintelligence/issue24/brackencontrol.html

It is necessary to **reduce nutrient inputs and soil fertility**, often with a complete stop to fertilizer application[54]. Some of the measures to reduce soil fertility listed below may also be needed eg top soil removal, sod cutting.

Scrub and tree removal can be carried out by cutting and chopping biomass with a flail or rotary mower, or cutting with a brush cutter (but leave some habitat structures out for birds, butterflies etc.). In some cases a grassland abandoned for 5 years can be restored by scrub clearance and an intensive restoration mowing (2 or 3 times per year). However, more often it is necessary to also control perennial weeds and invasive species, and scrub control must be repeated a number of times. Restoration effort differs considerably depending on speed of succession and invasion of weedy or non-native species. Nutrient-poor (and dry) grasslands tend to lose their biodiversity value more slowly but nutrient-rich sites degrade more rapidly as a result of the growth of dominant grass species, scrub development, and invasive species.

The restoration of hydrological conditions for wet grassland is vital to the success of restoration. It may include measures to block ditches to restore groundwater levels or large-scale measures to redesign flood management by removing or modifying dikes, sluice gates etc., so that grassland can be flooded periodically or to restore groundwater dynamics.

Degraded grassland may require **re-seeding** in patches or into the sward, which may entail seed bed preparation (ploughing, disking), spreading seed-rich hay from suitable grasslands (green hay or dried hay), or buying or gathering native seed (gathered and use within certain regions), and either bedding in seeds with roller on bare soil patches or sowing seed into the sward using a slot-seeder. Sow seed manually on small patches. Cultivated grass varieties should not be used in ecological grassland restoration (Conrad and Tischew, 2011). It is necessary to control weeds and invasive species, particularly to ensure special management for the first vegetation period after species transfer on productive sites. Intensive 'shock' grazing or mowing can reduce rank vegetation and reduce competing weeds. Sheep grazing from late winter to early spring is sometimes considered to control ragwort and other perennial weeds, and all-year-grazing is also effective in controlling dominant (invasive) species (eg *Calamagrostis epigejos*).

Re-creation of species rich grassland

Re-creation of species rich grassland requires some or all of the following measures:

Land must be cleared of previous land use (if it is not arable land), eg clear-cutting of forestry plantation, clearance of scrub and/or trees. Soil fertility can be reduced by, eg growing a nutrient hungry catch crop eg maize for several years, burying topsoil by deep ploughing, and removing top soil and turf. The seed bed may then need to be prepared by ploughing or disking.

The land may have retained some native seed bank, but generally an initial mix of semi-natural grassland propagules needs to be introduced through spreading hay from suitable grasslands (green hay, dried hay, seed-threshing or brushing material), or buying or

[54] NB this measure alone, when starting from eutrophic soil conditions, is not sufficient to restore mesotrophic plant communities such as fen meadows, but has the potential to restore plant communities of species rich hay meadows and pastures (such as Lolio-Potentillion and Cynosurion cristati)

gathering native seeds if no donor populations for direct harvesting are available. It is important to bed in seed with a roller. After grassland creation it is necessary to control weeds and invasive species using the same kinds of measures as above. The establishment of yellow rattle (*Rhinanthus major*) to control dominance of grasses is sometimes recommended, but is not always reliable or effective.

Measures to restore hydrological conditions for wet grassland types are similar to the restoration measures above, but may be more laborious and expensive, and generally need to be linked to a more large-scale habitat recreation project, including ditch blocking or restoration, measures to raise the groundwater table linked to floodplain restoration, and buffer zones to prevent the grassland becoming degraded by agricultural run-off.

Additional measures may be undertaken to provide small habitat features within grassland for specific species, eg the creation of shallow water scrapes in rough grassland, or hedge planting.

Source-targeted measures such as decreasing groundwater abstraction and tackling nitrogen emissions (moving nitrogen sources like farms away from sensitive habitats, applying best available techniques, etc) can be used to reduce external pressures, but are not specifically related to semi-natural grasslands alone. These measures are dealt with in Chapter 5.

Key measures that address the key pressures on Europe's grasslands are given in Table 6-17. Many of these need to be carried out over all areas where they relate to specific pressures. However, some depend on the type of grassland and its condition. Mostly importantly, it is considered that grazing is needed over virtually all semi-natural grasslands. Although some are traditionally only cut for hay, most meadows are also grazed for some part of the year. Therefore, although we do not have exact data on the proportion of semi-natural grasslands that are grazed, we assume that it is required over 90% of their area at some time.

As mentioned, above a high proportion of semi-natural grasslands are Annex 1 habitats, and of these meadow habitat types cover 2,209,225 ha of the total Annex I grassland habitat area, which amounts to about 28%. However, a review of management practices for such habitats shows that in practice some areas of some meadow habitats, such as tall humid herb grasslands (3% of area) and some *Molinia* meadows (2 % of area) are not routinely mown (Olmeda et al, 2013). Therefore, assuming that the proportion of Annex 1 habitats that are mown is broadly equivalent to the proportion of all semi-natural grasslands that are mown, we presume that mowing will be required over 25% of the ecosystem area.

Little information exists on the proportion of semi-natural grassland that will require proactive restoration measures, but it is reasonably likely that most areas will be able to recover from pressures through natural processes. However, intervention may be required on some areas, especially to achieve restoration by 2020. Information on the area over which pro-active measures will be required is generally lacking, and therefore the estimated percentage areas requiring scrub removal, soil fertility reductions and re-seeding are largely expert judgments / assumptions.

As discussed above, insufficient information exists to be able to quantify the extent of semi-natural grassland that is significantly impacted by alien invasive species, and therefore these measures are not included in the table or costed below.

Table 6-17 Key maintenance and restoration measures for permanent and semi-natural grasslands and their required percentage coverage to address each key pressure

Degradation source / key pressures	Extensive grazing*1	Traditional mowing for hay*1	Fire prevention and control	Manage / restore farmland features	Hydrological management	Reduce soil fertility	Clear scrub or tree cover	Restore hydrological conditions	Sow / reseed (in establishing windows)	Recreate habitat (incl all measures)
Ecosystem loss										100
Inadequate management or abandonment	90	25	?				10			
Grassland intensification	90	25				5			10	
Hydrological modification and drainage					100			100	10	
Inappropriate burning or wildfires	(50)		100						1	
Loss of habitat features				100						20
Weeds and invasive species									?	

Notes: *1 These measures should also include limits on the use of artificial and organic fertilisers, lime and other agrochemicals.

6.4.6 Costs of maintenance and restoration measures

The costs of maintain and restoring grasslands vary mainly according to grassland type (with specific management requirements) and grassland topography and ease of access, eg in relation to the use of machinery. Significant restoration costs relate to the restoration and management of hydrology (eg fen meadows) and the removal of scrub or trees.

Costs data for maintenance were sourced from the majority of Member States[55], mainly from their agri-environment programmes, but it should be noted that large areas of grazing may be on unregistered common land or 'forest' land not in receipt of CAP payments. As discussed in section 3.4, agri-environment payments often do not reflect the full costs of

[55] AT, BE, BG, CZ, DE, DK, ES, FI, FR, HU, IT, LT, LV, NL, PL, SE, SK, UK

maintenance of these habitats, particularly for the most valuable biodiverse grasslands from which little to no market-based income can be obtained due to their low productivity. Where these habitats are still managed, farmer and shepherd incomes are made up of a combination of CAP direct payments and a package of CAP Rural Development payments including Less Favoured Area payments (Poláková et al, 2011).

From an analysis of all the costs data, the following conclusions can be drawn:

Combined maintenance costs
The Rural Land Use Costs Study (Hart et al, 2011) reviewed agri-environmental payments in 21 different Rural Development Programmes, and found that general management measures (eg extensive grazing, mowing, reduced inorganic fertiliser inputs and measures for certain species) for all grassland types had an average payment rate of 230 €/ha/yr across 121 different management options. But payment rates varied greatly, from a minimum payment level of 7 €/ha/yr in the Netherlands to a maximum of 1,103 €/ha/yr in Denmark. However, this average is for all grasslands, and the maintenance of specific valuable grassland habitat types often appears to require a higher budget. For example, standard costs in the Netherlands range from 80 to over 2,000 €/ha/yr for different habitat types (Verheijen et al, 2009b). Wet grassland types have higher costs for maintenance of ditches and water level control infrastructure. Management of wet meadows and pastures for species diversity and/or wintering birds in the Netherlands is costed at around 500 €/ha/yr, management of flower rich wet meadows is costed at over 1,600 €/ha/yr, and species rich nutrient poor wet grassland is the most expensive to maintain, including 1,262 €/ha/yr for mowing with a low compaction vehicle.

Therefore taking into account the apparent higher cost for some semi-natural grasslands, in this study we use a slightly higher cost than the average used in the Rural Land Use Costs Study, and assume an average cost for **combined management of 250 €/ha/yr**.

Costs of specific measures:

Grazing: Specific agri-environment payments for grazing on important biodiverse grasslands include

- 90 to 195 €/ha for grazing alpine meadows with transhumance of livestock in Austria (Keenleyside et al, 2012)
- grazing species-rich *Nardus stricta* grassland is paid in Sweden at 120-270 €/ha/yr (depending on nature conservation value of the land); in Poland at 205 €/ha/yr (Galvánek and Janák, 2008)
- 308 €/ha/yr for managed grazing of dry steppe grassland and heathland in the Czech Republic (Keenleyside et al, 2012); 22-200 €/ha/yr for grazing pseudo steppe in Spain (San Miguel, 2008)
- 76 €/ha/yr for extensive grazing in France (Keenleyside et al, 2012)
- 2 €/ha/yr for open grazing of cattle and 4 €/ha/yr for managing deer grazing on rough open range grassland in Scotland (The Scottish Government, 2012)
- 30 €/ha/yr for grazing floodplain grassland in Scotland, UK (The Scottish Government, 2012)

- shepherded grazing of dry calcareous grassland in Italy Trentino province is paid at from 50 to 100 €/ha/yr (shepherded milk cow grazing with 1 herder for 50 LU; shepherded heifer grazing with 1 herder for 80 LU; shepherded sheep grazing with 1 herder for 100 LU); but in a LIFE project for dry calcareous grassland, rotational grazing on easily accessed low to moderate elevation areas, with sheep and equines, cost from 125 to 200 €/ha/yr (Calaciura and Spinelli, 2008a). The agri-environment payments do not cover the full costs of shepherded grazing. Shepherded grazing for nature conservation in Saxony-Anhalt, Germany, is costed at 450€/ha (Landesregierung Sachsen-Anhalt, 2009).

In addition to costs there are often additional costs for the maintenance / renewal of fences and water supplies, and these are often costed and paid separately. Such costs vary greatly according to the type and remoteness of the terrain. Fencing costs are usually quoted per metre rather than per hectare, but some example per hectare payments include 15 €/ha for fixed fences to over 500 €/ha for electric fencing of dry calcareous grassland in Italy (Calaciura & Spinelli, 2008a). Fencing costs were estimated at 750 €/ha and water supply costs at 2063 €/ha on lowland calcareous grassland in the UK (GHK, 2006). LIFE projects have spent 135 to 1150 €/ha in grazing infrastructure, including fencing, drinking equipment, corrals, shelters, and 13 €/ha for water supply (Calaciura & Spinelli, 2008a). However, it is not possible to establish or reliably estimate for this study the area of grasslands that needs fencing or the length of fencing required within these areas. Therefore fencing costs are not included in this study.

On the basis of the available agri-environment and other more specific cost data, this study assumes an average cost for grazing maintenance (excluding fencing costs) of **150 €/ha/yr**.

Mowing: Specific agri-environment payments for mowing of ecologically important grasslands include:

- 75 €/ha/yr for mowing of mountain hay meadows (fixed mowing date at least once a year with removal of cuttings; limit to manure application) in the Czech Republic (Keenleyside et al, 2012)
- Mowing of mountain hay meadows in Austria – 350 €/ha/yr for mowing with tractor; 430 €/ha/yr for mowing with motor mower; 700 €/ha/yr for mowing by hand) (Keenleyside et al, 2012); mowing of mountain hay meadows in Germany – 290-370 €/ha/yr, with an additional 100 €/ha/yr for restrictions on fertiliser use in Natura 2000 sites (Landesregierung Sachsen-Anhalt, 2009)
- 120 to 375 €/ha/y for mowing species-rich *Nardus stricta* grassland in Sweden (depending on nature conservation value of the land) (Galvánek & Janák, 2008)
- 308 €/ha/yr for mowing of alluvial meadows (Molinion and Cnidion litter meadows) in Poland (Šeffer et al, 2008); 150 to 170 €/ha/yr for mowing of alluvial meadows (Molinion and Cnidion litter meadows) in Slovakia (combined AEM and LFA payment (Šeffer et al, 2008); 320-400 €/ha/yr for mowing *Molinia* meadows and 180 €/ha/yr for mowing alluvial Cnidion meadows in Germany (Landesregierung Sachsen-Anhalt, 2009)
- 183 to 202 €/ha/yr for mowing of meadows with particular nesting bird species in Czech Republic (payments vary according to types of birds present) (Keenleyside et al, 2012) and 450 €/ha/yr for the same in Germany (Thomas et al, 2009)

- 220 to 260 €/ha/yr for mowing by tractor and removal of hay on dry natural grassland (rupicolous pannonic grasslands) in Hungary, or 110 €/ha/yr if the cuttings are left lying (LIFE 08 NAT/H/000288)

This study assumes an average cost for **mowing maintenance of 200 €/ha/yr**.

Information on the costs of **fire prevention and control measures** for grasslands is not readily available, but basic measures such as the provision of signs and basic fire control equipment are likely to be relatively low compared to other costs. Furthermore, the costs of supporting measures relating to legislation, enforcement and awareness raising are not included in this study. For this study fire prevention and control measures are assumed to be **5 €/ha**.

The maintenance of farmland features is a requirement under cross-compliance regulations and therefore its cost are considered to fall on the landowner, rather than to be supported through agri-environment measures. However, in this study it is considered that the management of **farmland features** may receive agri-environment, or other public funding, where the actions go beyond the retention of the features and aim to enhance and restore them. The costs of such actions are taken to be the same as those for all farmland boundary features which were found in the Rural Land Use Costs study to average **154 €/ha**.

Information on the costs of **hydrological maintenance** are lacking for most countries. However, data from the Netherlands indicate that the cost of maintaining ditches on wet grassland is between 38 to 100 €/ha, and needs to be carried out every two years (Verheijen et al, 2009b). This study therefore assumes an average maintenance cost for hydrology of wet grasslands of **35 €/ha/yr**.

Combined restoration costs
Overall costs depend greatly on the type of grassland being restored, particularly if restoration and management of ditches and water level control features is needed. Consequently, previous reviews have found that general restoration costs of existing grasslands vary widely, from 100 to over 3,000 €/ha/y (Eriksson, 2008; GHK, 2006). Funded measures typically include the establishment of grazing and/or mowing with removal of biomass, fencing and water supply installation, scrub clearance, bracken or thistle control, and species translocation.

This current study found that combined measures cost between 50 €/ha and 7,000 €/ha. However, the latter was for the restoration of alluvial flood meadows in Austria (Šeffer et al, 2008) and is probably not typical, as the next highest cost was 2,131 €/ha, for the restoration of upland meadows in England. Excluding the Austrian figure gives an average of **622 €/ha**, which we therefore use in this study.

Costs of specific key restoration measures:

The **reduction of soil fertility** normally requires the removal of the top layers of soil, and such physical works are therefore always very costly. For example, according to the data collated for this study:

- Costs of top soil removal for fen meadow restoration were 16,700 to 26,000 €/ha in Poland (Klimkowska et al, 2010a; Török et al, 2011)
- Top soil removal (with or without vegetation) for grassland re-establishment cost an average of 6,245 €/ha (range 2,574 – 10,126 €/ha) in four restoration areas within two LIFE projects in France.

A less expensive alternative can be adding sawdust and glucose to the soil, as this can effectively reduce soil fertility of sandy arable soils and reduce weeds, and only cost 4,000 €/ha (8 applications per year for 3 years) (Török et al, 2000). But we found no evidence that this technique is widely adopted.

On the basis of a rounded average of the above costs of top soil removal we assume the one-off cost of **soil fertility reduction is 14,000 €/ha**.

Scrub removal costs vary considerably, primarily according to the degree the degree of scrub development. For example:

- Scrub removal after more than 10 years of abandonment cost on average 3,237 €/ha on nine sandy or calcareous grassland different sites, and an average of 1,540 €/ha on two different hay meadow sites in Germany (Conrady et al, 2012)
- Agri-environment payments for scrub removal on dry calcareous grassland (Festuco-Brometalia) in Italy vary from 60 to 3,000 €/ha, depending on whether machinery can be used or how remote the terrain is (Calaciura & Spinelli, 2008a).
- Scrub clearance costs in LIFE projects ranged from 140 to 53,800 €/ha. For encroachment rates of between 50% and 75% costs still ranged from 550 to 5,000 €/ha, so costs are very site specific.

The extreme cost of 53,800 €/ha appears to be atypical, and more importantly it would not seem appropriate to restore such sites to meet Target 2, as clearly other sites could be restored at much lower cost. Therefore, we exclude this outlier cost, and assume an average one-off restoration cost for **scrub control of 3,350 €/ha**.

Abandoned grasslands may require **restoration grazing**, using for example breeds of livestock that are able to graze and browse low quality forage, such as tall rank vegetation and invasive species (eg as bracken), whilst at the same time trampling and breaking up scrub. This normally requires the use of traditional hardy breeds, which are not as profitable as modern breeds or as widely available. Therefore the cost of obtaining and maintaining livestock for restoration grazing is often higher than that of normal maintenance grazing. For example, **restoration grazing costs averaged 1,447 €/ha** (between 281 and 3,061 €/ha) in three different LIFE projects, for the first five years from restoration (Herremans et al, 2013).

Restoration mowing may also be carried out to remove built up biomass and to allow grazing and subsequent hay prediction. Similarly restoration mowing is usually more expensive than normal mowing because it often needs to be carried out by hand or the use of specialist equipment. Consequently three LIFE projects were found to have restoration mowing costs varied of 76, 281 and 2,165 €/ha (Herremans et al, 2013). For this study we therefore take the rounded average of these, ie **840 €/ha** as the usual cost of restoration mowing.

Little information appears to be available on the specific cost of **hydrological restoration** measures. However, the overall costs of the restoration of alluvial floodplain meadows in the Morava-Dyje floodplain project was almost 7,000 €/ha on the Austrian side, 536 €/ha on the Slovak side (Neuhauser 2001 quoted in (Šeffer et al, 2008). Restoration of boreal alluvial meadows cost 500-600 €/ha (LIFE03NAT/S/000070) (Eriksson, 2008). It is difficult to deduce an average cost for the hydrological component of restoration from this information, but for the study we assume one-off hydrological restoration of **500 €/ha** for wet grassland types.

Where degradation has resulted in the loss of the original vegetation and impacts on the seedbank, such that natural regeneration is not possible, then some re-seeding of vegetation will be required. Example cost data for such actions include:

- Re-seeding by seed mixtures or hay transfer cost 170 to 2,500 €/ha in a range of LIFE projects.
- Re-seeding by hand is costed at 14-18 €/ha treated and re-seeding by slit seeding into the sward at 10-18 €/ha treated in Germany (Bavaria), assuming that the patches needing treatment are only 0.01 ha in size (Bayerisches Landesamt für Umwelt, 2011).
- plant material transfer for fen meadow restoration cost 425 €/ha in Poland (Klimkowska et al, 2010b; Török et al, 2011)

Generally the hay transfer method is both the most cost-effective and the most ecologically appropriate method, provided that local hay from the target grassland type can be sourced. Therefore this study assumes an average cost of **1,000 €/ha/yr** for re-seeding of degraded grasslands, where possible through hay transfer. However, it should be noted that re-seeding does not normally need to be carried out over the entire area, but typically over 10% of the degraded area.

The cost of restoration of **farmland features** is taken to be the same as the management and restoration of hedgerow features, although other features types (such as scrub, walls, ponds and trees) may be restored. These costs were found in the Rural Land Use Costs study to average **503 €/ha**.

Control of invasive species, weeds and scrub (including weeds, scrub and trees). Costs obviously vary greatly according to the infestation rate and the persistence of the species. Control is reimbursed by agri-environment payment in England (UK) at 60 £/ha (75 €/ha) (Natural England, 2009), but actual costs for control of invasive species and persistent weeds on restored habitat sites under LIFE are from 65 to 3,000 €/ha, an average of 900 €/ha, with the highest costs for manual removal of invasive trees and prevention of regrowth (Herremans et al, 2013). In contrast, in the Netherlands it is calculated that control of invasive species on 5% of a habitat area will cost just 6 €/ha/yr (Verheijen et al, 2009b). Costs for control of *Asclepias syriaca* ranged from 290 to 900 €/ha/y of infested habitat in Hungary, depending on the method used.

Given the vary wide ranges in costs for the treatment of invasive species and uncertainty over the area that they significantly impact, no attempt is made in this study to estimate an overall cost for these measures.

Habitat creation costs

This study found fourteen examples of the combined costs of semi-natural grassland re-creation on arable farmland and improved grasslands, which ranged between 235 and 3,270 €/ha, and averaged 1,175 €/ha. However, it is likely that these costs do not include all necessary activities, for example soil preparation or scrub removal. On the other hand, the examples mostly relate to small-scale projects and costs may be lower for larger scale initiatives.

Re-creation activities that generate significant costs include deep ploughing or top soil removal in order to reduce soil fertility, and sourcing of species-rich or highly specific seed mixtures. Agri-environment payments for 11 different options for grassland and wetland creation in the UK average 490 €/ha, with a minimum payment level of 88 €/ha in England and 1,168 €/ha in Scotland (Hart et al, 2011). Other payments for grassland creation on arable land in the UK, France, Finland, and the Czech Republic range from 170 to 2,000 €/ha, depending on the seed mix (Keenleyside et al, 2012; Pywell et al, 2002; The Scottish Government, 2012; Török et al, 2011). Seeding dry natural grassland (rupicolous pannonic grasslands) cost 1,000-1,420 €/ha in Hungary, including ploughing soil for reseeding at 220-250 €/ha, gathering or purchasing seed at 430-770 €/ha, and seeding by hand at 400 €/ha (LIFE 08 NAT/H/000288). Grassland creation is funded at 193-245 €/ha for sowing seed by machine, 486-653 €/ha by hay transfer, or 1,046-1,328 €/ha by hydroseeding in Germany (Bavaria) (Bayerisches Landesamt für Umwelt, 2011). Seeding with native seed mixtures including site preparation is costed at an average of 3,300 €/ha in Thuringen, Germany (Freistaat Thüringen, 2003). Removal of topsoil for grassland re-establishment cost 1,800 to 10,100 €/ha in LIFE projects. Restoration mowing and hay removal cost 275 €/ha/yr for the first three years after rupicolous pannonic grassland re-creation on arable land in Hungary (Halassy and Török, 2004).

Taking all the varying evidence into account, we consider that the combined cost estimates (which average 1,175 €/ha) may not be comprehensive and therefore we assume an average one-off cost of **2,000 €/ha for grassland habitat re-creation from arable land**.

Costs for grassland creation from forestry plantations include these costs plus the costs of tree removal. In some situations grassland re-creation may follow on form the commercial harvesting of trees. But often tree growth on former semi-natural grasslands is poor and the harvesting of trees is not profitable, especially if they are to be felled before maturity. Furthermore, tree stumps and brash need to be removed from most or all of the area, following felling. Consequently the removal of trees is often an activity that needs to be at least partly subsidised. In fact an assessment of LIFE projects shows that tree removal and other associated actions to prepare for seeding, can be expensive, with costs varying between 552 and 10,000 €/ha (Herremans et al, 2013). It is, however, likely that the 10,000 €/ha figure is not typical, and therefore we do not take an simple average but assume the removal of trees costs **4,000 €/ha.**

This study therefore assumes an average one-off cost of **6,000 €/ha for grassland habitat creation from forestry.**

It is very likely that most grassland re-creation is likely to take place on arable or improved grasslands, rather than on forested land. Therefore, assuming that 75% is on farmland we adopt a **combined re-creation cost of 3,000 €/ha.**

6.4.7 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration and re-creation targets. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). Two estimates are provided of the costs of maintaining the ecosystem (ie preventing further losses and degradation) and restoring and re-creating 15% of the area of the ecosystem that has been degraded since the baseline year, based on two methods: firstly the overall level of ecosystem degradation and average combined costs of maintenance and restoration (see to Table 6-18), and secondly, the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures (see Table 6-19). See section 3.4 for a detailed account of the methodology and its assumptions, and section 10.1 for a discussion of the results of the cost estimation in relation to previous studies.

The estimated additional annual costs of **maintaining** the ecological condition of semi-natural grasslands is between €81 to 488 million, according to overall degradation rates and combined measures costs, or between €43 to 178 million based on key pressures and the specific costs of key measures. The results from the two methods provide cost estimates that are in the same magnitude, but the maximum estimate (ie based on the highest estimate of degradation rates) is much higher for the estimate based on overall degradation rates and combined measures costs. As discussed in the chapter 3, the estimates based on key pressures and key measures are expected to be more precise and reliable. It is therefore expected that the possible maximum cost is closer to €178 million, rather than €488 million.

The estimated average additional annual costs between 2010 and 2020 of achieving the **restoration target** for semi-natural grasslands is between €70 to 231 million, according to overall degradation rates and combined measures costs, or between €54 to 505 million based on key pressures and the specific costs of key measures. These results show reasonable consistency for the lower degradation level estimate, but the estimate based on key pressures and key measures is twice as great as that based on general degradation rates and combined measures. Also it should be noted that there is a ten-fold difference between the lower and upper estimates for the latter method. These differences are largely due to the high levels of uncertainty regarding the requirements for proactive management measures to overcome the effects of abandonment.

The estimated average additional annual costs between 2010 to 2020 of achieving the **re-creation target** for semi-natural grasslands is between €45 to 83 million, according to overall degradation rates and combined measures costs, or between €28 to 54 million based on key pressures and the specific costs of key measures.

Table 6-18 Projected additional costs (€) in 2020 of achieving Target 2 in semi-natural grassland based on the basis of the current costs of combined measures and expected overall degradation extent in 2020

Important notes: Costs do NOT include supporting actions, wide-scale / at source measures to address pollution impacts, or species-specific costs. All costs are gross costs in Euros and based on current values using cost estimates from as close to 2012 as possible. Costs and totals are rounded to three significant figures. Costs are all additional to existing (baseline) measures, and expected measures (ie the reference scenario) up to 2020. The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. The ecosystem area = 32,550,311 ha.

Activity	Cost	Area required over		Cost in 2020	
	€/ha	Min	Max	Min	Max
Maintenance required in 2020	250	1.00%	6.00%	81,400,000	488000,000
Restoration required 2010-2020	622	3.45%	11.4%	69,800,00	231,000,000
Re-creation required 2010-2020	3,000	0.44%	0.86%	43,400,000	83,500,000

Table 6-19 Projected additional costs (€) in 2020 of achieving Target 2 within semi-natural grassland on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

Important notes: All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. See section 3.5 for further explanation. Ecosystem area = 32,550,311 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min	Max					Min	Max
ANNUAL ONGOING MAINTENANCE NEEDS IN 2020								
Abandonment and under management	0.200%	1.000%	Extensive grazing	150	90%	AEM	8,790,000	43,900,000
	0.200%	1.000%	Mowing	200	25%	AEM	3,260,000	16,300,000
Overgrazing	0.000%	0.000%				GAEC		
Grassland management intensification	0.500%	1.800%	Mowing	200	90%	AEM	29,300,000	105,000,000
	0.500%	1.800%	Farmland habitat feature protection		100%	GEAC	-	-
	0.500%	1.800%	Farmland habitat feature management	154	6%	AEM	1,500,000	5,410,000
Hydrological modification	0.100%	1.000%	Maintenance of hydrology	35	8%	AEM	91,100	911,000
Inappropriate burning and wildfires	?	?	Fire prevention and control					
Loss of habitat features	0.100%	2.000%	Farmland habitat feature protection		100%	GEAC	-	-
	0.100%	2.000%	Farmland habitat feature management	154	6%	AEM	301,000	6,020,000
Invasive species	?	?						

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instru ment	Annual costs in 2020	
	Min	Max					Min	Max
							-	-
MAINTENANCE TOTAL							***43,200,000***	***178,000,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% Area requiring restoration in 2020		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Abandonment and under management	0.750%	8.750%	Restoration grazing	1,450	90%	AEM	31,900,000	372,000,000
	0.750%	8.750%	Restoration mowing	840	10%	AEM	2,050,000	23,900,000
	0.750%	8.750%	Scrub clearance	3,350	10%	AEM	8,180,000	95,400,000
Over-grazing	0.000%	0.000%				GAEC	-	-
Grassland management intensification	0.000%	0.000%	Reduce soil fertility	14,000	5%	AEM	-	-
	0.000%	0.000%	Reseeding	1,000	10%	AEM	-	-
Hydrological modification	0.750%	0.900%	Hydrological restoration	500	100%	AEM	12,200,000	14,600,000
Inappropriate burning and wildfires	?	?	Reseeding	1,000	1%	AEM	-	-
Loss of habitat features	3.000%	9.000%	Restoration of hedgerows etc	503		AEM	-	-
Invasive species	?	?				AEM		
RESTORATION TOTAL							***54,300,000***	***506,000,000***

ADDITIONAL RE-CREATION NEEDS TO REVERSE CONVERSION SINCE 2000

Key Pressure	% area requiring re-creation in 2020		Key measure	Sub measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max						Min	Max
Conversion to forestry	0.002%	0.011%	Re-creation	From forest	4,000.00	100%	AEM	195,000	1,460,00
Conversion to crops	0.420%	0.810%	Re-creation	From crops or improved grassland	2,000.00	100%	AEM	27,300,000	52,700,000
RE-CREATION TOTAL								***27,500,000***	***54,200,000***

6.5 FOREST

6.5.1 Description and extent

According to the CLC typology, forest includes all areas where the vegetation cover is composed principally of trees, including shrub and bush understories. The typology distinguishes between broad-leaved forest (CLC 311), coniferous forest (CLC 312), mixed forest (CLC 313) and transitional woodland scrub (CLC 324). The forest area includes both naturally regenerated and planted forest with a canopy cover of at least 30%, including wooded or afforested dunes, bogs, fens or wetlands. Transitional woodland-scrub mainly represents areas of either degraded or regenerating woodland.

'Broadleaved forest' is forest where the canopy of broadleaved species represents more than 75% of the total surface area; 'coniferous forest' is forest where coniferous species represent at least 75% of the total surface area; and 'mixed forest' is where neither broad-leaved nor coniferous species occupy more than 75%. 'Transitional woodland scrub' includes areas of bushy or herbaceous vegetation with trees over more than 30% but less than 50% of the area. Regenerating but still open burnt or clear cut forest areas, heavily damaged forest areas with more than 50% dead trees, and forest nurseries and young plantations are also classified as transitional woodland scrub. Transitional woodland scrub is a transitional habitat between forest and grassland or heath or scrub habitat, and cannot therefore be considered "restorable" in the same sense as the other land cover areas in this report[56]. This study therefore disaggregates the total transitional woodland scrub area amongst the forest, semi-natural grassland, heathland and sclerophyllous scrub ecosystems, according to the proportions of each in each Member State (see annex 1 for more details).

Forest covers at least 35% of the EU land area. A summary of the area of broadleaved, coniferous and mixed forest habitat in each Member State is provided in

Table 6-20. This study uses the total area of 151.8 million ha forest plus woodland-scrub in 2006, 40% of which was predominantly coniferous forest, 25% predominantly broad-leaved forest, 20% mixed forest, and 15% transitional woodland-scrub. Large recently burnt areas inside forest areas (CLC 334 burnt areas) covered another 110,073 ha, mainly in Mediterranean countries. For comparison, Member States reported a total of 177 million ha of forest and wooded land in 2010 (Forest Europe et al, 2011). Whilst the forest area in Bulgaria, Denmark, and Ireland has been increasing at around 1-2% a year over the last five years, and shrinking by 0.44% in Estonia, the differences between the two statistics are most likely to be due to the limitations of the CLC land cover system (see section 3.4.1 for more explanation). There is also a very large discrepancy in the data sets between the area of forest in Spain. The CLC land cover is therefore an underestimation of the total area of forest, due to the effect of forest fragmentation, because isolated areas below 10 ha in size are not registered as forest. Despite the CLC data limitations, they are used here to provide consistency with the ecosystem types covered by this study. **Thus the total forest area is taken to be 151,940,126 ha.**

[56] The MAES Working Group is considering the exclusion of CLC 324 transitional woodland scrub from ecosystem mapping methodologies for Europe for this reason (Maes et al, 2012).

Table 6-20 Area of forest and share of transitional scrub CLC (hectares), and area of forest reported by Member States to FOREST EUROPE in 2010

The column shaded in grey denotes the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. The data for Greece are subject to uncertainties. Area of forest and other wooded land 2010 is what Member States reported to Forest Europe (Forest Europe et al, 2011).

Land cover (ha)	Broad-leaved forest CLC 311	Coniferous forest CLC 312	Mixed forest CLC 313	Share of transitional woodland-scrub CLC 324	Share of forest in farm mosaic areas CLC 243	Burnt areas CLC 334	TOTAL forest and forest-scrub and forest in farm mosaic areas	FOREST EUROPE: forest & wooded land 2010
Austria	408,850	2,186,711	1,122,394	15,668	91,959	35	3,825,617	3,991,000
Belgium	204,601	139,723	265,740	16,238	90,778		716,332	706,000
Bulgaria	2,305,087	533,872	637,744	596,114	151,700	370	4,224,887	3,927,000
Cyprus	747	153,477	358	17,903	14,379	245	187,109	387,000
Czech Republic	278,478	1,723,105	616,388	130,990	94,372		2,843,333	2,657,000
Denmark	73,424	180,017	135,564	67,197	57,434		513,636	635,000
Estonia	445,682	796,784	832,124	394,794	69,043		2,538,427	2,337,000
Finland	756,875	9,682,032	9,294,883	4,638,558	154,985		24,527,333	23,116,000
France	8,796,652	3,472,541	1,938,319	953,340	919,507	16,833	16,214,124	17,572,000
Germany	2,412,968	5,616,905	2,393,871	176,991	385,735		10,996,401	11,076,000
Greece*	4,969	949	901	1,424	166		8,409	6,539,000
Hungary	1,501,327	98,172	155,479	201,126	49,579		2,005,683	2,039,000
Ireland	29,095	227,160	29,757	195,597	73,288		554,896	787,000
Italy	5,468,194	1,284,991	1,089,223	782,045	525,923	2,841	9,153,217	10,916,000
Latvia	554,607	908,852	1,162,082	528,814	122,394		3,249,562	3,467,000
Lithuania	428,149	720,947	729,986	209,120	167,884	295	2,267,679	2,249,000
Luxemburg	63,850	11,846	17,840	585	8,658		102,779	88,000
Malta		67	140	-	2,014		2,221	350
Netherlands	59,865	161,014	94,132	968	82,210		398,300	365,000
Poland	1,508,685	5,583,484	2,316,327	270,462	358,094		10,037,052	9,319,000
Portugal	1,022,021	538,493	486,481	860,426	162,893	32,886	3,103,200	3,611,000

Romania	4,811,767	1,068,425	1,127,286	406,992	242,139		7,656,609	6,733,000
Slovakia	1,072,698	502,117	396,757	188,814	48,431		2,199,765	1,938,000
Slovenia	441,848	247,624	448,718	39,520	57,387		1,235,097	1,274,000
Spain	3,753,222	3,873,910	1,503,092	2,026,344	796,979	54,882	12,033,758	27,747,000
Sweden	2,001,877	21,000,403	1,664,148	4,450,151	85,626	1,686	29,203,891	30,625,000
United Kingdom	534,181	1,355,001	143,967	87,841	31,408		2,140,812	2,901,000
Total	**38,939,719**	**62,068,622**	**28,603,701**	**17,321,246**	**4,844,963**	**110,073**	**151,940,126**	**177,002,350**

Forest cover is unequally distributed in Europe, with most occurring in Sweden (17% of the EU total), Finland (13%), Spain (16%), France (10%), Germany and Italy (6%), Poland (5%), and Greece and Romania (4% each). The other Member States have 2% or less of the forest area. Native forest in Boreal Europe and mountainous regions is mainly coniferous or mixed, whilst Western and Central Europe has predominantly broad-leaved forest, with mixed forest in mountainous and transitional regions. Mediterranean forest contains broad-leaved and sclerophyllous elements, including native pine forests. However, all Member States have areas of mixed forest that are partly the result of forest planting during the last century (EEA, 2008).

Plantation forests cover around 12.9 million ha, which is 7% of the EU-27 forest area (see annex 1 for details of the definition). The proportion of plantation forestry to semi-natural and natural forest area varies greatly between Member States depending on the degree of historical deforestation and different forestry practices. Plantations make up most of the forest area in Ireland (89%), Denmark (77.5%) and the UK (77%). In Belgium (42%), Luxembourg (32%), Portugal (24.5%), Romania (22%), and Bulgaria (21%) plantations make up a third to a fifth of the forest area. Spain has 15% plantations, whilst the other Member States have less than 10% to no plantation forest. Spain and Portugal have seen a rapid increase in the area of forest plantations during recent decades, established mainly on formerly extensively grazed land (EEA, 2008).

Ecologically valuable forest habitats, as defined in Annex I of the Habitats Directive, cover about 27.4 million ha of the EU-25 (see annex 1 for more details). This is equivalent to around 21% of the CLC area, but does not include any of the substantial areas of forest in Romania and Bulgaria.

6.5.2 Key pressures and extent of degradation – 2010 baseline levels

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures at the baseline year, which for this study is taken to be 2010. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Ecosystem loss

CLC land cover change analysis[57] shows that between 2000 to 2006 forest and woodland-scrub expanded by 106,300 ha, which amounts to about 0.017% per year over the period (EEA, 2010a). This was partly as a result of the abandonment of grasslands in Southern Europe, but also due to an increase in plantations. As the total area of forest is increasing in the EU, overall habitat loss is not an issue; however, this hides the continuing loss of some areas of highly biodiverse forest, for example Białowieża in Poland, and declines in certain Member States, such as Estonia (Forest Europe et al, 2011). Fragmentation of forest habitat, often from transport infrastructure, is also an important issue that is not addressed in this study (EEA, 2009a).

Overall degradation – baseline

Member States reported to Forest Europe in 2010 on pressures affecting their forests (Forest Europe et al, 2011). Data were reported on 69% of the EU-27 forest area[58]. The largest proportions of damaged forest were reported for Portugal (24.5%) and Italy (22.5%), followed by Sweden (12.2%), Hungary (12.1%), Cyprus (5.8%) and Lithuania (4.7%). In the remaining 14 Member States the proportion of forests with damage ranged from 3.4% in Bulgaria to less than 0.1% in Finland. Because the forest area is so heavily skewed towards particular Member States and because damage surveying and reporting practices vary greatly, the average proportion of area reported as damaged by one or more pressures is only 6.44% (Forest Europe et al, 2011). This is likely to be too low due to under-reporting (Forest Europe et al, 2011).

Defoliation is an indicator of the general condition of trees, as a decrease in the biomass of photosynthesising needles or leaves affects the vital functions of trees and reduces their growth. The State of Europe's Forests 2011 reports a fifth of surveyed trees as damaged over the last five years (Forest Europe et al, 2011). In contrast, Finnish forests, which are considered to be relatively undamaged, had 6% of pine, 20% of spruce and 5% of broadleaves classified as damaged in 2009 (on non-peat soil) (Metla, 2012).

There is further evidence from condition monitoring under the Habitats Directive that the extent of forest degradation reported above significantly underestimates degradation in terms of biodiversity aspects. According to the last assessment 63% of Natura 2000 Annex I forest habitats (which comprise about 13% of EU forest area) are in unfavourable status, and the status of a further 16% is unknown (EEA, 2010a). In the Atlantic, Boreal and Macronesian regions no forest habitat types have an overall favourable status (EEA, 2010a).

---

[57] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

[58] Austria, Belgium, France, Greece, Spain, Luxembourg, and Malta did not supply information

As the Annex I habitat locations should correspond well with the CLC cover (see annex 1 for details), it can be assumed that the Article 17 assessments provide a reasonably representative and reliable indication of the level of degradation of the ecosystem type. However, there are some major gaps in coverage, including a lack of data about the extent of 23 forest habitat types (such as Mediterranean pine forest) in Spain, which has almost a fifth of the EU's Annex I forest habitat (ETC/BD, 2008a).

Taking into account these widely differing degradation estimates, this study assumes that baseline degradation levels may vary between **20%** and **45%**.

Specific key pressures - baseline

Intensive forestry management with high felling levels, even-aged stands, lack of old trees, and lack of dead wood

According to the EEA 87% of forests in EEA member and collaborating countries are subject to some degree of intervention (EEA, 2008). At its most extreme clear felling and replanting has a devastating effect on forest ecosystems, and is, for example, destroying areas of Mediterranean *Pinus nigra* forests, which are habitats of European Community Interest (Zaghi, 2008a). Other aspects of intensive forestry management also place significant pressure on forests almost everywhere, although forest management is showing a long-term improvement in some areas. Over-intensive exploitation results in the replacement of semi-natural forest habitats with monocultural plantations of one-age class and with a limited understorey. Clear felling contributes to habitat fragmentation and can also lead to rapid succession to dense shrub communities that prevent forest regeneration (EEA, 2010a). In addition, valuable habitat features such as ponds, wetlands, or stone walls are often removed or filled in to permit forest works.

Sustainable management of forests should involve adequate representation of associated species and formation of irregular structures, containing trees of various ages, including very old specimens in order to secure genetic variability. European forests, however, are largely even-aged: 12% of the forest is under 20 years old, and 42% is between 20 and 80 years old. Whilst 8% of the even-aged area can be ascribed to plantation forestry, the majority of the forest area is degraded because the older trees have all been removed. Only 18% of forest is over 80 years old (Forest Europe et al, 2011).

Sustainable management of forests should involve leaving an appropriate amount of dead wood. Dead wood, both in standing and lying trees and timber, is a key source of nutrients for living trees and an important habitat for numerous saproxylic beetles, birds, bats and mosses of special conservation value (Thauront and Stallegger, 2008). The proportion of dead and decaying trees – a key indicator of forest ecosystem health – is far too low in many European forests (EEA, 2010a). The average volume of standing and lying deadwood reported in forests in 2010 varied from 19-26 m^3/ha in Austria, Lithuania, Slovenia and Slovakia, to 6 m^3/ha or less in Denmark, Finland, Ireland, Poland, and the UK (Forest Europe et al, 2011)[59]. Ireland, Denmark, and the UK are dominated by even-aged stands of

[59] Ten Member States (Bulgaria, Cyprus, France, Greece, Hungary, Luxembourg, Malta, Portugal, Romania and Spain) did not report this data.

plantation forestry that are intentionally very low in dead wood. The forests of the other Member States are mainly semi-natural or natural, and the low volume indicates over-intensive exploitation and widespread burning of small wood and other debris. In Swedish forests, whole trees including twig and leaf biomass are removed for conversion to biomass energy production, removing substantial amounts of biomass and associated nutrients that would otherwise be left on the forest floor. A critical indicator of the proportion of deadwood for biodiversity is the status of saproxylic beetles: 14% of species are considered threatened (Nieto and Alexander, 2010).

Although the exact area degraded as a result of intensive forestry management is uncertain, there is good evidence that a wide area is affected in terms of age structure, proportion of dead and dying trees and of dead wood. Therefore, this study concludes that at least **60%** of the forest area is degraded as a result of intensive forestry management and, according to the EEA data noted above, the upper limit (with some rounding up) could be **90%**. The annual rate of degradation through management intensification in 2010 is more difficult to estimate, but given the high proportion that is now under intensive management it is likely to have declined over recent decades to low rates, and for this study we assume it could vary from **0.1% to 1%** per year.

Conversion to plantations dominated by non-native forestry species, including invasive species

A major threat to European forests, and especially broad-leaved woodlands (and other ecosystems such as heathlands and semi-natural grasslands) is clearance to establish coniferous plantations (Thauront & Stallegger, 2008). Introduced forestry species tend to have a much lower biodiversity value than native trees because far fewer species are adapted to live on them. Between 2000 and 2006, 94% of the change in forest cover in the EU was due to felling and transition, and conversion to commercial tree types (EEA, 2010a). Whilst in Ireland, Denmark and the UK commercial non-native tree species are clearly dominant (on 70% and 49% respectively) because of the predominance of plantation forestry, the forests of Belgium (42%), Hungary (35%), Luxembourg (30%), the Netherlands (25%), and Portugal (29%) are also significantly dominated by introduced tree species (Forest Europe et al, 2011). In contrast, other Member States report only 7% or less. Overall, Member States reported 19% of their forest area as dominated by introduced tree species; although this may be an underestimate as some Member States including Finland did not report this information[60]. For the purposes of this study we assume 20% to 30% of EU forest cover is dominated by non-native species.

It is unclear what proportion of the 7% of forest cover that is plantation forestry is converted from former natural or semi-natural forest. However, the proportion is likely to be relatively small as much afforestation is known to occur on poor quality arable land, semi-natural grasslands, heathlands and some peatlands, often after agricultural abandonment. Furthermore evidence from analysis of land cover changes from 1990 to 2000 indicate that overall forest area increased significantly over the period in many countries, averaging 6% across 24 countries (Büttner et al, 2006). In the absence of better

[60] Eight Member States (Estonia, Finland, Greece, Latvia, Lithuania, Malta, and Romania) did not report these data.

data we assume that only about 10% to 20% of forest plantations are converted forests. Thus, overall with some rounding **0.6% to 2.0%** of EU forest cover is assumed to be plantation forest on former natural or semi-natural forest. However, it is not possible with the available data to estimate the annual rate of conversion in 2010.

Invasive plant species

Invasive tree species were reported by 10 Member States as a problem in forests in 2010 (Forest Europe et al, 2011). Hungary and France reported that invasive tree species are a problem on 0.23% and 0.11% of their forest area respectively, but on average Member States reported only around 0.05% of their forest area as affected. However, this is over a fifth of all of the area these states report as dominated by introduced tree species, indicating that a significant proportion of introduced forestry species have become invasive, particularly *Eucalyptus* spp., *Robinia pseudacacia*, *Ailanthus altissima*, and *Prunus serotina*. The non-native *Pittosporum undulatum* tree species was introduced to the Azores and has significantly altered native Laurel forests (Guimaraes and Olmeda, 2008). Large areas of forest are also affected by invasive shrubs, in particular *Rhododendron* species (EEA, 2008).

This study assumes that **0.05%** to **0.2%** is degraded through the presence of invasive tree and other plant species.

Fire damage

Although burning can be beneficial in the management of some woodland habitats, particularly boreal forests (see below), uncontrolled fires that are too intense or frequent can lead to severe habitat damage and irreversible biodiversity loss. During 2006, EU Member States reported a total area of 312,886 ha of burnt forest (JRC, 2012)[61], which represents 0.21% of the CLC forest land cover area. In 2007 the burnt area was 0.42% of the CLC forest area. Wildfires in coniferous forest in the drier Mediterranean region are a particular threat because the dominant tree species such as *Pinus nigra* rely on annual seed recruitment, meaning a fire can destroy a whole generation of seeds and seedlings (Zaghi, 2008a). Native Laurel forests on the Canary Islands are highly threatened by wildfires (Guimaraes & Olmeda, 2008). A high frequency of forest fires in Mediterranean sclerophyllous forests can encourage the development of invasive herbaceous vegetation, such as the grass *Ampelodesmos mauritanica,* which provide large quantities of dry litter that further increase the frequency of fires[62]. Because wildfires changes the vegetation, they create more fire-prone forest habitat that is then more frequently revisited by fire (Moreira et al, 2011).

It is very difficult to estimate the area of forest that is at risk of fires, is burnt and is actually degraded due to forest fires, because the frequency of fires and their impacts vary greatly across the EU. Furthermore, some burnt areas will already be degraded by previous fires, and fire recovery periods will vary according to the forest type, the severity of the last fire and the frequency and impacts of previous fires. For this study we round up and down the 2006 and 2007 fire coverage data respectively and assume 0.2 to 0.45% of forest area was affected by fires each year in the baseline period. We also assume that on average burnt

[61] Eight Member States (Belgium, Denmark, Ireland, Luxembourg, Malta, Netherlands, Romania, UK) did not report any burnt area.

[62] http://www.sciencedirect.com/science/article/pii/S0378112700004357

forests will be significantly degraded for 20 to 50 years (although some could be for much longer or even permanently). This therefore leads to an assumed overall degradation extent of **4% to 22%.** The area at risk of damaging fires (ie outside the boreal region) and therefore requiring prevention and control measures, will be much greater but is uncertain.

Inadequate burning management

Fire is a natural and important ecological component of some forest types, and in particular boreal forest is currently suffering from a lack of disturbance due to a decline in natural wildfires (Forest Europe et al, 2011; Similä et al, 2012). Boreal forests comprise around 18% of the total forest area and we therefore assume that between **10%** and **18%** of the forest area is impacted a result of the cumulative effects of be reduced fires and therefore requires proactive burning restoration over a part of this area and then ongoing burning management.

Invertebrate damage and diseases, including invasive species

All trees are affected by herbivorous invertebrates and phytopathogens (bacteria, viruses, fungi) to some extent but major pest and disease outbreaks may cause major impacts on forests, resulting in a weakening of forest ecosystem health and vitality, and economic losses. Forests are more vulnerable if they are already affected by eutrophication or acidification, drought, or heavy storm damage. However, a certain level of pest damage can create valuable standing dead wood habitat and maintain a certain level of disturbance. The spread of insects and diseases is identified in the Forest Europe report 2011 as affecting 3.2% of the European forest area (Forest Europe et al, 2011). However, again Member States' reports vary widely from 20% of forest affected in Portugal and Romania, to 8% in Hungary and Italy, and 3% or less in most of the others who reported this information[63]. Insect and disease damage is often connected to other forms of forest damage, such as storm damage and air pollution, so it is difficult to estimate the overall level of degradation that can be attributed to it.

Boreal conifer forest trees are susceptible to large-scale damage from *Scleroderris* canker and resintop disease, and continuous damage is caused by *Annosus* root-rot, whilst there are periodic outbreaks of pine sawfly (Metla, 2012). Most damage, however, cannot be attributed to any identified causal agent. Mediterranean *Pinus nigra* forests are vulnerable to damage by caterpillars of the genus *Thaumetopoea* which cause intensive defoliation leading to reduced growth rates, particularly among younger plants (Zaghi, 2008a). New invasive alien invertebrate pests such as the Oak Lace Bug (*Corythucha arcuata*) are causing some major problems (EEA, 2008).

Given the likely under-reporting of insect damage by Member States, this study assumes that at least **3%** and possibly up to **10%** of the forest area is degraded due to insect and disease damage, including invasive pest and disease species.

Reduced forest regeneration due to wildlife or livestock

[63] Eight Member States (Austria, Belgium, France, Greece, Ireland, Malta, Netherlands, Spain) did not report this data.

Some grazing and browsing in forests can maintain species-rich grassland patches, and create open wood-pasture habitats that are often very important for biodiversity. However, over-intensive grazing and browsing by farmed sheep or cattle and/or intensive selective browsing by moose, deer or rabbits can significantly limit forest regeneration (Forestry Commission, 2000; Metla, 2012; Natural England, 2008; Thompson et al, 2003). This can lead to a loss of characteristic vegetation communities and habitat structure, as observed in Caledonian pine forests in Scotland (Perrow & Davy, 2002). In some countries, such as the UK, high native deer densities are causing significant impacts on forests, including very low rates of regeneration. Furthermore, this problem is often exacerbated by some invasive alien species, including Sika Deer (*Cervus nippon*) and Muntjac Deer (*Muntiacus reevesi*) (EEA, 2008). Impacts are most likely to be severe where livestock are grazed in natural forest habitats (ie not pastoral woodlands ect) as, in addition to grazing and browsing, significant additional impacts may arise from trampling and eutrophication, which can alter soil composition and structure, leading to a change in ecological communities.

The EU-27 Member States reported a 2.2% rate of forest damage due to wildlife (Forest Europe et al, 2011). However, it is not clear how much of this is significantly degraded the ecosystem, so this study assumes that **1 – 2%** of forest is annually degraded due to high levels of grazing and browsing by livestock and wild animals.

Hydrological modification and drainage

Forest intensification often includes the drainage of peatlands and wet forest. For example, Finland drained on average 200,000 ha of forest a year for two decades until the 1990s (Peltomaa, 2007). Land drainage has a particularly negative effect on biodiversity (EEA, 2010a). Riverine forests are degraded by changes to river flows through flood protection, dam building or channelization (Tucker & Evans, 1997) and almost 90% of alluvial forest has been lost in Europe (WWF, 2001). Drainage and other modifications to hydrological regimes are a major threat to coniferous forests such as Caledonian forests in Scotland, which are closely associated with wet peatland habitats (Perrow & Davy, 2002). Floodplain forests and mire and swamp forests cover at least 6% of the EU-27 forest area (EEA, 2006a) and these are especially susceptible to hydrological modification. See the section on peatlands for more discussion of bog woodland and the section on freshwater ecosystems for more discussion of alluvial forests.

Currently the area degraded by hydrological modifications is uncertain, but is clearly significant. In the absence of better information, it is assumed that between **1%** and **4%** of forest area is degraded due to hydrological modifications. There is inadequate EU level information available to provide a reasonable estimate of the annual rate of hydrological modification in 2010.

Soil degradation and erosion in forests

Soil degradation is a problem in intensively managed forests from roads and skid trails, which are prone to erosion and soil compaction. Intensive timber harvesting with large machinery results in soil compaction, which decreases water infiltration and root penetration. Heavy shading, dropping of acidic needles and the use of heavy machinery during timber harvesting result in signification changes to soil structure and composition, which can hinder the recovery of native broadleaved species if, or when, plantations are removed (Thauront & Stallegger, 2008). However, 20.5% of the EU-27 forest area was reported as being maintained for the protection of soil, water and other ecosystem services (Forest Europe et al, 2011). It can be assumed that on much of this area, management plans and/or legal restrictions at least reduce to some extent soil degradation through human activities.

Currently the extent of soil degradation in forests is uncertain, but it is likely that it is restricted to plantation forests (~10% of forest cover) and the most intensively managed semi-natural forests. For this study we therefore assume that the extent of degradation is between **5%** and **10%** of forest area. There is inadequate EU level information available to provide a reasonable estimate of the annual rate of soil degradation in 2010.

Under-management of woodlands
Some woodlands, such as in the UK, are affected by under-management and as a result have a low age and structural diversity, and lack clearings which are often important for many woodland species, such as butterflies and other insects. Such problems tend to affect woodlands that were clear-felled (often during the Second World War) and naturally regenerated. Many formerly coppiced woodlands are also no longer managed, resulting in overly dense and even aged stands.

Information on the extent of under-management is lacking, but it is unlikely to affect the large areas of high forest in Scandinavia, and central and eastern Europe, and is most likely to affect smaller woodlands in western Europe where management is uneconomic. It is therefore assumed that only **1% to 5%** of Europe is degraded by the cumulative impacts of under-management. The annual rate of degradation from the abandonment of management of woodland management in 2010 is difficult to estimate, so a wide range of **0.1% to 0.5%** is used in this study.

6.5.3 Key pressures and extent of degradation in 2020

It is likely that the forest area in Europe will continue to increase until 2020. Because European forests are on balance actively growing, their stock of sequestered carbon in above and below-ground biomass is increasing by 1.25% a year, providing an important ecosystem service (Forest Europe et al, 2011). The long-term trend for Europe's forests is towards greater tree species diversity, indicating greater overall biodiversity and proportion of native species (Forest Europe et al, 2011).

Currently, EU forests are not particularly well protected with only some 13% within Natura 2000 sites as of 2007 (EEA, 2008), although 20% of the EU-27 forest area is subject to less stringent protection through felling restrictions for biodiversity and landscape protection (Forest Europe et al, 2011). This includes areas with no active intervention, with minimum intervention, and with conservation through active management. On average, in this area, 52% of the wood volume can potentially be felled (Verkerk et al, 2008). This study assumes that the area of forest with felling restrictions will increase from the baseline level to 30% by 2020[64], and that in this area intensive management will cease or be better regulated so that the age distribution and the proportion of dead wood will improve by 2020. It is also

[64] for example Sweden planned to protect another 900,000 ha of forest by 2010

possible that Natura 2000 sites may have better management in future in line with European Commission guidance (European Commission, 2003). However, use of funding for biodiversity conservation management provided from the CAP through Natura payments or other forest measures is currently very low, and may well decline as a result of proposed cuts to the CAP (see section 8.2).

In the remaining area, a moderate increase in timber removal from EU-27 forests to 2020 can be compatible with an increase in deadwood. But an increase in the use of forestry residues for energy production from biomass is expected and this could fully counteract this development and lead to a significant reduction in deadwood (Verkerk et al, 2011). The type of deadwood would also change, as greater proportions of smaller deadwood fractions (ie stem residues) are removed, as well as a decrease can be expected in the amount of standing deadwood and of large-diameter deadwood. Without additional management measures to protect deadwood, intensification of biomass removal could significantly increase the negative effect on deadwood-dependent species by 2020 (Verkerk et al, 2011).

It is important to note that climate change will put Europe's forests under increased pressure from summer drought and heat waves, and significantly increased damage from forest fires, storms, winds and abrupt heavy snow fall (EC, 2010). Insect and disease damage is likely to increase with increasing climate change (EC, 2010). There seems to have been a decrease in wildfire damage in the last few years, but climate change is predicted to significantly increase the likelihood of wildfire damage over the next two decades (Ramón Vallejo et al, 2012).

Weighing up the net effects of these impacts from these changes in drivers and policies is difficult. However, it seems there is little evidence to suggest that there will be significant changes in most pressures up to 2020, though uncertainty in the estimates of overall and key pressure-specific degradation will increase. This study therefore assumes that average degradation levels will not change but adopts a slighter wider range of minimum and maximum levels, as summarised in Table 6-21. The only exceptions are for impacts from fires and invertebrates and diseases, which will probably continue to increase due to the spread of existing diseases exacerbated by climate change. In contrast under management may decline as a result of increased management for biomass.

Table 6-21 provides an indication of the proportion of Europe's forests affected by each pressure at current baseline levels and as expected in 2020 according to the reference scenario.

Table 6-21 Estimated baseline and expected 2020 percentage degradation levels (a) and conversion rates (b), and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020	
	Min %/y	Max %/y	Min %/y	Max %/y
Overall	**11.2%**	**21.5%**	**11.1%**	**21.9%**
Intensification of forest management	0.100%	1.00%	0.050%	1.50%
Conversion to plantation	?	?	?	?
Fire damage	?	?	Increasing ?	Increasing ?
Inadequate burning management	10.0%	18.0%	10.0%	18.0%
Reduced regeneration due to wildlife damage and/or livestock	1.00%	2.00%	1.00%	2.20%
Hydrological modification and drainage	?	?	?	?
Soil degradation	?	?	?	?
Abandoned traditional management	0.100%	0.500%	0.050%	0.250%

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels		Maximum level for 15% restoration		Expected in 2020		% restoration required in 2020		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	10.0%	40.0%	8.5%	34.0%	8.0%	48.0%	0.00%	14.0%	
Intensification of forest management	60.0%	90.0%	51.0%	76.5%	60.0%	95.0%	9.0%	18.5%	
Conversion to plantation	0.600%	2.00%	0.510%	1.70%	0.600%	2.20%	0.090%	0.500%	
Invasive alien species	0.050%	0.200%	0.043%	0.170%	0.100%	0.500%	0.058%	0.330%	
Fire damage	4.00%	22.0%	3.40%	18.7%	5.00%	25.0%	1.60%	6.30%	
Inadequate burning management	10.0%	18.0%	8.50%	15.3%	10.0%	18.0%	1.50%	2.70%	
Insects and diseases	3.00%	10.0%	2.55%	8.50%	3.00%	12.0%	0.450%	3.50%	
Reduced regeneration due to wildlife / livestock	1.00%	2.00%	0.850%	1.70%	1.00%	2.20%	0.150%	0.500%	1
Hydrological modification and drainage	1.00%	4.00%	0.850%	3.40%	1.00%	4.40%	0.150%	1.00%	
Soil degradation	5.00%	10.0%	4.25%	8.50%	5.00%	12.0%	0.750%	3.50%	
Abandoned traditional management	1.00%	5.00%	0.850%	4.25%	0.500%	5.00%	0.000%	0.750%	

Note: 1 Area includes significant remaining impacts from fires up to 50 years ago

6.5.4 Key management and restoration measures

The management of forests for biodiversity and related ecosystem services is complex and much has been written on the subject. It is not the intention of this report to provide a detailed or comprehensive review of possible forest management objectives or actions, or to identify all possible measures. Instead the section below draws on a number of key references (Ausden, 2007; EEA, 2008; European Commission, 2003; Fischer et al, 2012; Perrow & Davy, 2002; Peterken, 1993; Similä et al, 2012) and identifies the most important actions that would potentially have significant costs.

One of the most important and widely required key measures forests will be the adoption of more **sustainable forestry management** practices that maintain or restore forest species composition diversity, structural diversity and other important habitat features. Thus felling or large areas should normally be avoided (although the creation of some large open areas may be beneficial in certain situations). Standing and fallen deadwood, should be retained as much as possible, and the majority of tree stumps left after harvesting. Dead wood is of particular importance and should be maintained at a density of approximately 5-15 m^3/ha in an irregular distribution in order to provide habitat for fungi, invertebrates and roosting bats (Thauront & Stallegger, 2008). An adequate number of trees, and patches of late successional forest, should also be left to go through their natural life cycle, and all veteran trees should be left as intact as possible, as these provide very important habitats for birds, bats and many invertebrates as well as fungi, mosses and lichens.

Planting should be mixed, and ideally only of native species (or at very least be dominated by native species).

Intensive forest management, especially ploughing and planting of new plantations, leads to significant soil damage. This can be mitigated by the use of appropriate low ground-pressure machinery to plant and harvest timber, (or even horses to remove timber) careful timing of forest actions (eg to avoid wet periods), working in small areas, and where necessary and appropriate the creation of protected tracks and drainage systems.

Reintroduction of traditional forest management

Where former traditional management of forests (eg by coppicing) has been abandoned then it will usually be necessary to thin overly dense stands, create spatial and structural variability, eg by reintroducing coppicing or pollarding in some areas, and possible clearance of trees to create permanent grassy clearings, and unshaded ponds and ditches etc.

Removal of non-native trees

Where native forests have been replaced with commercial non-native plantations, the introduced trees will need to be cleared. Gradual clearing is usually recommended in order to prevent rapid succession in the cleared patches resulting in dense shrub communities that will restrict tree regrowth (Guimaraes & Olmeda, 2008). This may be achieved by the felling of successive patches, or by selective thinning or 'shelterwood felling' where occasional mature trees are retained to protect recovering seedlings. To save costs, felling of plantations can be timed to coincide with timetabled harvesting periods (Perrow & Davy, 2002). This 'continuous cover forestry' reduces disturbance to fauna, allows control of vegetation development, provides shelter for recovering plants, maintains microclimates and promotes gradual development of woodland favouring recovery of a wider range of age classes and species (Thompson et al, 2003). The method used will depend on economics, site conditions and which tree species is the target of restoration; beech, for example, recovers well under shelterwoods but cannot tolerate the high light and temperature of clear cut sites. In some sites, costs and access may prevent the use of gradual clearance techniques and clear-felling may have to be carried out. In all cases of clearance, any native trees within the woodland mix should be retained and encouraged whilst non-native species, particularly if they are invasive, should be targeted for removal (Thompson et al, 2003).

Replanting of native and genetically local species

It is generally recommended that recovery should be by natural regeneration wherever possible, for both ecological and economic reasons (Thauront & Stallegger, 2008; Thompson et al, 2003). However, it is critical that the causes of habitat loss, and any other agents restricting seedling growth, are first removed. In some cases, where the seed bank of native trees is lost or natural recovery slow for other reasons, replanting of characteristic tree and understorey species may be necessary. Seeds or seedlings should be native and preferably local, and to reduce the risk of genetic pollution, it is important to avoid planting trees of unknown origin in the proximity of autochthonous woods as intraspecific hybridisation can easily occur among different subspecies of native tree species such as pines or oaks. The pattern of planting should mimic natural regeneration as far as possible, in an irregular pattern and maintaining a mix of species with different growth rates (Perrow & Davy, 2002; Thompson et al, 2003). The soil may need to prepared, and 'ripping' (upturning soil along contour lines to create furrows) is a less expensive and often more effective option than creating individual holes for planting (Zaghi, 2008a).

Control of over grazing by wildlife or livestock and regeneration of native understorey

High grazing pressure, usually by deer, can threaten woodland regeneration. Direct browsing of leaves and shoots suppresses seedling growth, whilst trampling and dunging can alter soil composition and structure. Where grazing rates are very high, all seedlings may be eliminated, but where the pressure is intermediate, species more susceptible to grazing will be disproportionately affected leading to a change in species composition, which may last several decades if the woodland is allowed to mature. Protection of restoration areas from grazing may be achieved by using plastic tree guards to protect individual trees, or by erecting metal or wooden fencing around the whole site. Fencing will need to be maintained over several years to prevent grazing whilst trees mature (Löf et al, 2010). Regenerating conifer seedlings are vulnerable to preferential grazing, which suppresses their recovery, and in some areas culling is used to reduce deer browsing pressure (Perrow & Davy, 2002). Alternatively, shrub species can be retained or planted to provide shelter for regenerating seedlings (Zaghi, 2008a).

Full restoration of woodland habitats includes the recovery of characteristic ground flora and understorey communities. In areas where the seed bank has persisted, and grazing / browsing pressures are low, natural regeneration can be encouraged by successive clearing of patches (Thompson et al, 2003). On the other hand, felling can result in a rapid expansion of a dense understorey that can crowd out regenerating trees. Hand-cutting is least damaging to wildlife but herbicides are usually less expensive and more rapidly effective

(Thompson et al, 2003). Carefully controlled light grazing can also be an effective method of managing understorey development to favour regeneration of species associated with the habitat, and at the same time to reduce the risk of fire (Thauront & Stallegger, 2008; Zaghi, 2008a).

Control of insect pests and diseases

In areas affected by insect pests, the pest species should be controlled through aerial and/or ground chemical treatments, and/or by indirect measures such as thinning, cleaning, pruning and weeding, plus the use of tree species resistant to insect attack. Long term management through thinning, pruning and weeding may also be necessary to strengthen trees' resilience (Zaghi, 2008a). Finland has a policy of strict control of timber storage in the summer, and this is considered to have significantly decreased pest and disease damage (Metla, 2012).

Fire prevention and control

The complete prevention of damaging forest fires is not possible (or in some cases not desirable, as described below), but measures can be taken to reduce their frequency and to aid their control if they should breakout. Such measures include, the provision of warning signs, regulation of activities of at vulnerable sites during periods of high fire risk. The creation of fire breaks and the provision of basic fire control equipment can aid the control of fires if they do occur.

Controlled burning

For some forest types, fire is an important ecological component and it can therefore be beneficial to intentionally reintroduce controlled fires into forest management in appropriate circumstances, in particular where it is safe to do so. Fire is therefore a recommended forest management measure in boreal forests and is used to create habitats for fire-dependent species, increase the amount of deadwood, affect the quality of wood in living trees and diversify the structure of forest (Similä et al, 2012). It can therefore be an important tool for restoring the forests that have low structural diversity and deadwood volumes, as result of previous clear-felling and/or intensive management.

It is, however required infrequently, and only over a small proportion of the forest to create structural diversity.

Removal of invasive alien species

Restoration of woodlands affected by invasive alien species will require their removal or at least control. Mapping and inventory of the species will often be a critical first step, followed by eradication through the use of chemicals or mechanical removal. The latter may be more suitable on difficult terrain and for species which can regenerate from fragments, eg *Fallopia japonica* (Guimaraes & Olmeda, 2008), but is more expensive and labour intensive.

Hydrological restoration

In wet woodland habitats, restoration may require the reinstatement of more natural hydrological regimes through actions such as the removal of drainage channels and/or the construction of water control structures (Thauront & Stallegger, 2008).

Table 6-22 indicates where each key measure needs to be used to address each pressure. All key forest measures are required over the entire area (here 100%), with the exception of replanting and controlled burning. The requirement for controlled burning will vary locally in different forest types and areas, but is expected to be on average required over only about 10% of degraded areas. Natural regeneration is likely to be feasible in most areas and therefore replanting is only expected to be necessary on a fairly small proportion of land, which estimate here to be about 5% of intensively managed non-plantation forests, or the 10% of burnt forests.

Table 6-22 Key maintenance and restoration and measures for forest and their required percentage coverage to address each key pressure

Degradation source / key pressures	Sustainable forest management	Removal of non-native trees	Removal of invasive alien species	Replanting of native species*	Re-introduction of forest management	Control of grazing by wildlife and livestock	Pest and disease control	Fire prevention & control	Controlled burning	Restoration of hydrology
Intensification of forest management	100			5						
Conversion to plantations		100		100						
Invasive alien species			100							
Fire damage				10				100		
Inadequate burning management									5	
Insects and disease							100			
Wildlife and livestock grazing						100				
Hydrological modification and drainage										100
Soil degradation	(100)									
Inadequate management / cessation of traditional management					100					

*Natural regeneration and planting may be used in combination, or a whole area may be recovered by either regeneration or planting

6.5.5 Costs of maintenance and restoration measures

As previously identified in this report, relatively few detailed data on habitat restoration costs are available in published literature. Where reports of woodland restoration involve an assessment of outcomes, the focus is usually limited to ecological impacts and any mention of costs tend to be non-monetary comparisons of the relative cost effectiveness of a restoration measure to alternatives.

Maintenance costs - combined

Appropriate key measures to maintain forests and prevent their degradation primarily relate to sustainable forest management practices and the control of certain risks as necessary (such as insect pest and disease outbreaks) and as such may often be carried out as part of good practice or to comply with regulations or sustainability standards. Therefore direct costs (ie excluding opportunity costs) of basic sustainable management of forests are likely to be normally negligible, or even of long-term commercial benefit. However, more onerous management measures, including avoiding clear-cutting large areas, avoiding thinning that is not necessary for biodiversity conservation management, and retention of all deadwood is likely to reduce profitability and therefore require compensation payments. According to Hart et al (2011) the average RDP compensation payment rate for restricting such forestry activities is **116 €/ha/yr**, and we therefore also use this figure in this study.

Maintenance costs – specific key measures

Inadequate data are available to estimate the individual costs of specific management measures, such **thinning, coppicing, pollarding, fencing and grazing**. Therefore the combined costs of management, as described above, of **116 €/ha/yr** are used in this study for basic forest management measures that aim to prevent intensification and deal with the impacts of grazing and browsing by livestock and wild herbivores.

Information on the costs of more **specific traditional management measures** are also generally lacking, but in the UK coppicing is estimated to cost 380 €/ha/yr (Verheijen et al, 2009b) and this figure minus the basic management cost of 116 €/ha/yr is therefore used here, giving an additional costs of **264 €/ha/yr.**

It should be noted that many **fire prevention and control measures** are already in place, especially in commercial forests and the area over which additional measures would be needed in 2020 is very difficult to predict, as well as their typical costs. However, it is likely that most would be borne by the forestry sector and would be relatively low compared to other maintenance management costs.

Restoration costs - combined

Most of the costs of achieving Target 2 for forest ecosystems are likely to be associated with restoration or re-creation. In particular widespread restoration will be necessary to overcome degradation from intensive forest management. A wide variety of studies have documented forest restoration costs, which show that although variable these cost are typically relatively high compared to some other ecosystems. For example, some costs for broadleaved woodlands are summarised below:

- A GHK report puts the capital cost of restoring native woodland in the UK at 3,000 £/ha (around 3,750 €/ha) (GHK, 2006).
- Restoration of a 'whole plantation' (ie trees, understorey, ground flora) is estimated by (Gilbert and Anderson, 1998) at 10,000 – 20,000 £/ha (around 12,500 – 25,000 €/ha), with the cost of restoring ground flora from seed at 1,000 £/ha (around 1250 €/ha) for a one-off operation
- The Woodland Trust calculates re-establishing broadleaved oak and birch woodland using a combination of natural regeneration and planting to cost 1,680 £/ha (around 2100 €/ha) (Pryor and Jackson, 2002)
- From the same Woodland Trust study, natural regeneration using 'shelterwood' cutting (retaining some mature trees to shelter seedlings) and deer fencing is calculated to cost 2,430 £/ha (around 3,037 €/ha) (Pryor & Jackson, 2002).

Costs of coniferous woodland restoration are rarely available in the literature. An overall cost of €597,170 is given for a LIFE project to restore native *Pinus nigra* forests in the Noroeste region of Murcia in Spain, involving clearance of competitive *Pinus* species, pruning of trees across 197 ha and planting of *P. nigra* on 28 ha (Zaghi, 2008). However, it is difficult to find what area of habitat is restored for this cost, and an itemised list of costs for individual measures is not provided, limiting the opportunity to apply the data more widely.

In Andalusia, Spain, the LIFE project LIFE03 NAT/E/000054 carried out the following measures to restore native pine forest on a total of 62 ha, at a total cost of €89,933 (corresponding to 1,450 €/ha):

- manual planting of native species on 21.3 ha, with a density of 800 plants per ha;
- reforestation of *Pinus pinea* forest on 21 ha with a plantation density of 1,000 plants per ha;
- removal of *Eucalyptus globulus* on 6.7 ha;
- pruning of 9.5 ha of young *Pinus pinea*; and
- preventative silvicultural treatments in high density pine forest, ie cleaning up of 4.9 ha through selective pruning of brushes with clearing machine.

In the UK under the Forestry Commission Scotland's Woodland Grant Scheme, landowners can receive funding for the restoration of Caledonian (Scots pine) woodland. Prices paid in 1998 were 1,350 £/ha for new planting on land areas smaller than 10 ha. With increasing land area, costs per ha were reduced to 1,050 £/ha for planting over 10 ha or more. Alternatively, rather than direct planting, land owners may choose to encourage restoration through natural colonisation, for which the Forestry Commission paid 525 £/ha plus 50% of the agreed cost for work to encourage natural colonisation. However, as the author notes, the ecological value of the restored forests varies considerably so it is difficult to assess to what degree restoration has been effective (Macmillan et al, 1998).

Despite these high documented costs the average payment available for forest conservation and restoration under RDPs is 133 €/ha, with a minimum of 36 €/ha paid in Hungary to a maximum of 268 €/ha paid in Denmark (Hart et al, 2011).

The average cost of the most typical combined restoration measures is 4,300 €/ha. It is clearly much greater than the average cost of the RDP measures, but is probably close to the real costs of more demanding and complete restoration. In contrast the RDP costs probably reflect less demanding restoration, such as increasing structural diversity and deadwood provision (see below). The average of **4,300 €/ha** is therefore used in this study.

Restoration costs - specific key measures

In some situations the **removal of non-native trees species** that have been planted as commercial crops may incur costs if they are being removed before they have grown sufficiently to produce an economically viable crop (which is especially likely on mires and moorland). However, to achieve Target 2, it need only be necessary to restore forests after alien tree species are removed as part of the commercial forestry cycle, albeit before maximum profitability may be achieved in some cases. This key measure is therefore considered to have negligible costs.

Costs for **planting native species** in the Netherlands vary from 1.2 €/ha for restoration of wooded dunes which require only 2% of their area to be planted for recovery of the whole hectare, to 4.7 **€/ha** to plant 9% of the land area to restore productive dry forests, to 35 €/ha for planting over 90% of the land area in park woodlands plus an additional 61 €/ha to plant ground flora bulbs across 20% of the area. In areas where planting is necessary to restore native vegetation, costs includes 0.34 €/ha for the purchase and planting of seedlings over 2% of the hectare (€17 if applied to the whole hectare) and 0.06 €/ha for application of manure fertiliser to 2% of the land area (€3 for whole hectare) (Verheijen et al, 2009b).

However, these costs are very low compared to other estimates, such as restoration projects with substantial planting components described above. Costs of planting according to information from LIFE projects in the UK, France and Italy averaged 1,880 €/ha. These estimates probably include the costs of associated actions (eg ground preparation, tree protection and fencing). On the other hand, a project to restore birch and rowan woodland habitats in the upland reach of the Pumlumon River in Wales, UK, provides estimated costs of 89 £/ha (ie approximately 110 €/ha) on average for tree planting including ground preparation and after-care, plus fencing at a costs of 553 £/km (Wildlife Trusts Wales, 2011). It is important to remember that regenerating seedlings will need protection from browsing animals, but calculating average fencing needs is very difficult, as information on cost on a unit area basis is lacking and needs are likely to be highly variable. Fencing costs are not therefore included as an additional specific cost, but they are likely to be relatively low compared to other planting costs, and often included in quoted planting costs anyway.

Therefore, taking these varying cost estimates into account, we assume the full typical cost of tree planting and associated activities, is closer to the estimates from the LIFE projects, and therefore taken to be **1,000 €/ha.**

The costs of **removal of invasive alien plant** species will vary considerably depending on the species involved, its density and the type and location of the forest habitat impacted. But removal cost are typically high for forests, and average **2,265 €/ha** according to data from LIFE projects in France, Ireland and Spain.

Information on the costs of **pest and disease control** are generally lacking, but this does not affect the estimation of the cost of achieving Target 2 as it is assumed that such measures will be normally be taken as part of commercial forestry activities. Outside commercial forests disease outbreaks would normally be expected to be left untreated unless they have severe impacts on the ecosystem. Where major potentially catastrophic disease outbreaks (such as Ash die-back) occur these will require wider action than beyond commercial forests and such cost are likely to be extremely high. But importantly, such measures would be expected to be taken anyway in the absence of Target 2. Pest and disease control costs, although very difficult to quantify for 2020, are therefore considered to be part of the reference scenario, and therefore not included in this study.

Specific costs of **hydrological restoration** for a number of broadleaved woodland types in the Netherlands are given in Verheijen (2009b). For example, bog woodland restoration in areas degraded by drainage will require reinstatement of water levels, estimated to cost 7.5 €/ha for activities on 80% of the area, which is sufficient to recover the whole area. Detailed data for hydrological restoration are lacking for other countries, and therefore the figure of **7.5 €/ha** is used in this study.

Taking these estimates into account, the assumed costs of specific key measures for forest restoration and re-creation are provided in Table 6-23 below.

6.5.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration target. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). Two estimates are provided based on two methods: firstly the overall level of ecosystem degradation and average combined costs of maintenance and restoration, and secondly, the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures. See section 3.4 for a detailed account of the methodology and its assumptions.

The results provided in Table 6-23 indicate that the estimated additional annual costs of **maintaining** the ecological condition of forest ecosystems is between €1,960 to 3,870 million according to overall degradation rates and combined measures costs, or between €196 to 488 million based on key pressures and the specific costs of key measures. These widely differing estimates reflect the high levels of uncertainty over degradation levels and the rather crude basis of the estimates using overall degradation levels and combined measures. Therefore, whilst it is difficult to provide reasonably precise estimates, it is likely that maintenance costs would be closer to the €196 to 488 million than those based on the other methodology.

The estimated average additional annual cost between 2010 and 2020 of meeting the **restoration target** for forests is between zero Euro (if expected restoration under the reference scenario is sufficient to achieve Target 2) and €9,150 million, according to overall degradation rates and combined measures costs, or between €140 to 519 million based on key pressures and the specific costs of key measures. As for the maintenance estimate, these widely differing estimates reflect the high levels of uncertainty over degradation levels and the rather crude methodology based on overall degradation levels and combined measures. These results should therefore be interpreted with great caution, especially the higher cost estimates.

As noted above, forest area has increased since the baseline year of 2000 and therefore no forest re-creation is required to achieve Target 2.

Table 6-23 Projected additional costs (€) in 2020 of achieving Target 2 in forest ecosystems on the basis of the current costs of combined measures and expected overall degradation extent in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. See section 3.5 for further explanation. Ecosystem area = 151,830,053 ha.

Activity	Cost €/ha	Area required over		Cost in 2020*	
		Min	Max	Min	Max
Maintenance required in 2020*2	116	11.10%	21.95%	1,960,000,000	3,870,000,000
Restoration required 2010-2020	4,300	0.00%	14.00%	-	9,150,000,000
Re-creation required 2010-2020		0.00%	0.00%	-	-

Note: * For restoration the cost is averaged over 2010-2020

Table 6-24 Projected additional costs (€) in 2020 of achieving Target 2 within forest ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). Duplicate costs are removed where "Below" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 151,830,053 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area affected in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs in 2020	
	Min/Y	Max/Y					Min	Max
Intensification of forest management	0.05%	1.50%	Sustainable forest management	116	100%	Below	-	-
Conversion to plantation	?	?	Regulation			Other	-	-
Fire damage	Increasing ?	Increasing ?	Fire prevention & control		100%	Commercial action	-	-
Reduced regeneration due to wildlife damage and/or livestock	1.00%	2.20%	Control of grazing / deer culling	116	100%	AEM / other	176,000,000	388,000,000
Hydrological modification and drainage	?	?	Regulation / Sustainable forest management			above	-	-
Soil degradation	?	?	Regulation / Sustainable forest management			above	-	-
Abandoned traditional management	0.05%	0.25%	Reintroduction of management	264	100%	AEM / other	20,100,000	100,000,000
MAINTENANCE TOTAL							***196,000,000***	***488,000,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% requiring restoration in 2020		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Intensification of forest management	9.00%	18.50%	Planting	1,000	5%	AEM / LIFE	68,400,000	141,000,00
Conversion to plantation	0.09%	0.50%	Removal of trees	-	100%	Commercial action	-	-
	0.09%	0.50%	Planting	1,000	100%	AEM /other	13,700,000	76,000,000
Invasive alien species	0.06%	0.33%	Removal of invasive species	2,265	100%	AEM / LIFE	19,800,000	114,000,000
Fire damage	1.60%	6.30%	Planting	1,000	10%	AEM/LIFE	24,300,000	95,700,000
Inadequate burning management	1.50%	2.70%	Controlled burning	10	5%	other	114,000	205,000
Insects and diseases	0.45%	3.50%	Pest and disease control	-	100%	Commercial action	-	-
Hydrological modification and drainage	0.15%	1.00%	Restoration of hydrology	8	100%	AEM/LIFE	171,000	1,140,000
Soil degradation	0.75%	3.50%	Sustainable forest management	116	100%	AEM /Certification	13,200,000	61,700,000
Abandoned traditional management	0.00%	0.75%	Reintroduction of management	264	100%	AEM/LIFE	-	30,100,000
RESTORATION TOTAL							***140,000,000***	***519,000,000***

6.6 HEATHLAND AND TUNDRA

6.6.1 Description and extent

This ecosystem type mainly comprises CLC type 322, referred to as 'moors and heathland'. This is generally described as vegetation with low and closed cover dominated by bushes, shrubs and herbaceous plants (heather, briars, broom, gorse, laburnum, etc.). The CLC types comprise Atlantic heaths (upland heaths often being referred to as moors), subalpine heaths and vegetated tundra. Atlantic heaths are secondary formations resulting from the transformation of forests through the combined impacts of tree felling, fires and grazing. They are also usually associated with shallow mineral soils. They comprise tall growth heaths composed of European gorse, bracken, etc, and low growth heaths composed mainly of Ericaceae heath. Subalpine heaths are formations based on rhododendrons, bilberries and *Calluna*, generally succeeding subalpine forest and grazing land. Heathland occurs in an ecological continuum with natural grassland (CLC 321) and forest (CLC 311, 312 or 313), and it is often difficult to distinguish them as separate CLC types. Wet heathland vegetation may develop on partially drained peat soils that were originally blanket bog or raised bog. These areas are discussed further in section 6.7. Tundra consists of dwarf shrub vegetation, sedges and grasses, mosses, and lichens. Scattered trees grow in some tundra.

This ecosystem also includes some areas of transitional woodland-scrub area (CLC type 324), which is regarded as including abandoned successional heathland, and is divided up among four CLC land cover types according to their proportion in each Member State (see annex 1 for details).

Annex I Natura 2000 habitats associated with the ecosystem type include 17 vegetation types - four dune heathland vegetation types, four dry heath types, three wet heaths, three alpine and boreal/sub-arctic heath and scrub types, and three South-east European scrub types (see annex 1 for details). The dune heathland vegetation types may be partially included in the beaches and dunes CLC type 331.

The areas of heathland and tundra ecosystems in each Member State are provided in Table 6-25. Sweden and the UK each have around a third of the EU-27's existing heath and tundra, and Spain has 12%. Finland, France, Portugal, and Austria have smaller amounts (between 7 and 4% each). Sweden, the UK and Spain each also have large areas of transitional woodland-scrub, and a share of the transitional woodland-scrub area has been added to the heath area for each Member State, in the assumption that some of the transitional woodland-scrub area is abandoned heath (see annex 1 for more detail on this calculation).

Sweden has large areas of tundra with boreal heath, the UK's area is dominated by wet heath with *Erica* species and dry heath with *Calluna*, and Spain's by gorse heath. Some of Sweden's heathland is picked up as sclerophyllous vegetation as it is dominated by *Juniperus* scrub (see section 6.7).

This study assumes the area of heathland and tundra is 8,949,182 ha, according to the above interpretation of CLC data.

Table 6-25 Area of heathland and tundra and share of transitional woodland-scrub CLC (hectares), and equivalent Natura 2000 habitat types

The column shaded in grey denotes the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. See annex 1 for details of Annex I habitat types that correspond to CLC types 322 and 324. Natura 2000 habitat area does not include any areas in Romania and Bulgaria.

Land cover type and Member State	CLC 322 Moors and heathland[1]	CLC 324 transitional woodland-scrub (share)	Total heath and tundra	Natura 2000 heath and tundra habitats (EU-25)
Austria	245,005	1,033	246,038	754,200
Belgium	16,107	409	16,516	11,872
Bulgaria	31,637	5,424	37,061	n/a
Cyprus		-	-	
Czech Republic	1,908	95	2,003	3,753
Denmark	50,076	8,650	58,726	47,300
Estonia	16,225	3,088	19,313	530
Finland	415,379	97,638	513,017	673,250
France	388,888	29,295	418,183	205,050
Germany	55,460	995	56,455	80,391
*Greece**	*523*	*109*	*632*	*150,198*
Hungary		-	-	1,270
Ireland	55,356	37,857	93,213	693,581
Italy	144,359	14,395	158,754	133,400
Latvia		-	-	1,429
Lithuania	3,362	394	3,756	2,270
Luxemburg		-	-	33
Malta		-	-	0
Netherlands	37,948	130	38,078	41,710
Poland	3,758	108	3,866	24,178
Portugal	292,528	122,960	415,488	100,790
Romania	70,851	4,115	74,966	n/a
Spain	14,042	1,280	15,322	3,883,157
Slovakia	22,524	782	23,306	17,611
Slovenia	934,208	209,928	1,144,136	36,600
Sweden	2,751,348	496,380	3,247,728	1,910,460
United Kingdom	2,277,215	85,409	2,362,624	1,114,617
Total	**7,828,707**	**1,046,222**	**8,949,182**	**9,888,150**

Note: [1] This land class also includes vegetated tundra.

6.6.2 Key pressures and extent of degradation – 2010 baseline level

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures at the baseline year, which for this study is taken to be 2010. See chapter 3 for a discussion of the estimation of baseline levels and available data.

Overall degradation levels – baseline

According to the EEA Baseline Report, 60% of Annex I listed heath and scrub habitats have an unfavourable status, and the status of a further 26% are unknown (EEA, 2010a). However, the EEA include sclerophyllous scrub habitats in their analysis, and therefore the results do not provide a reliable assessment of the overall status of this ecosystem type as defined in this study. Member States reported 30% of heath and tundra habitats in unfavourable status in 2007, and 38% in unknown status (ETC/BD, 2008a). The monitoring of these Annex I habitat types is generally good with no major gaps in coverage and a significant proportion of the heath, tundra and scrub ecosystems comprise Annex I habitats (Table 6-25). Therefore the Article 17 assessments probably provide a reasonably representative and reliable indication of the overall level of degradation of the ecosystem type. Consequently, with some rounding of the lower estimate, **overall baseline degradation are assumed in this study to be between 10 – 40% of ecosystem area** (but see further discussion below regarding air pollution impacts).

Specific key pressures – baseline

The key pressures affecting heath and scrub ecosystems and their associated species are from land-use changes related to conversion to agriculture or land abandonment, as well as pollution from atmospheric nitrogen (causing eutrophication) and air-borne deposition of sulphur dioxide (causing acidification) (Dise, 2011; EEA, 2010a). Air pollution pressures are described in section 5.5 whilst other key pressures are further described and quantified below.

Agricultural abandonment and undergrazing

Many heathland habitats are dependent on some grazing and/or burning (see below) as a means of preventing natural succession to forests. Appropriate grazing also helps to maintain structural and species composition diversity, and often creates more diverse habitats that burning or grazing (Backshall et al, 2001; English Nature, 2005; García et al, 2012; Gimingham et al, 1979; Gimingham, 1992). Consequently, the decline and loss of traditional grazing practices in the 20th century has been cited as a major cause of the loss of heathland vegetation throughout Europe (Newton et al, 2009). As a result of reforms of the CAP, livestock levels in most upland areas have fallen considerably in Europe over recent decades and upland heaths are now at risk of under grazing in many areas (Keenleyside & Tucker, 2010). This has allowed the natural succession of unmanaged scrub into the forest ecosystems, resulting in major declines in heath and scrublands (Vandewalle et al, 2010). Fragmentation and habitat loss may affect characteristic fauna species like the Sand Lizard (*Lacerta agilis*) (Schuna, 1999) and induce the need for active management as some varieties of heathland can only survive without active management when they cover large areas (Bal et al, 2001).

The primary drivers of agricultural abandonment and reduced grazing of moorlands are economic but other contributory factors include social drivers of rural depopulation especially in upland and remote areas (see Box 6.1). In contrast, in some countries, such as the UK, many lowland heaths are close to urban areas and as a result farmers are often unable to carry out grazing in such as locations due to animal welfare concerns. Also many heaths are now too small and fragmented to make grazing practical.

As described in Box 6.1 recent studies have suggested that annual abandonment rates were about 0.17% in France and 0.8% in Spain in the 1990s. Abandonment estimates for other countries with heathland and specific data on abandonment of heathlands is lacking, but it is known that these habitats are especially vulnerable to abandonment and undergrazing is frequently listed as a cause of unfavourable conservation status in Annex I habitats (EEA, 2010a). Therefore, we use the existing data as a guide but with such uncertainty assume a relatively wide range of possible abandonment rates, from between **0.1% to 1%.**

Abandoned heathland and moorland areas may remain as the same ecosystem type for a long time as succession to scrub and forest can be very slow. In fact succession can be extremely slow on moorlands that are remote from areas of scrub or tree seed sources, and heathlands that are prone to fire. Therefore we assume here that on average abandonment impacts last 30 years before they result in a change to forest ecosystems and the rate of abandonment has been relatively constant over the last 30 years. On this basis we assume the cumulative under-grazing and abandonment impacts now affect some **3% to 30%** of moorland and heathland ecosystems.

Over-grazing

Although, as described above, grazing of many of these ecosystems is beneficial for nature conservation, they can also be very sensitive to high grazing pressures, due to their low productivity and fragile soils. Such impacts can also be exacerbated by wild fires and inappropriate burning (see below). High grazing rates can result in changes in vegetation composition, to communities that are dominated by grazing resistant plants (García et al, 2012). This encourages an evolution towards grassland ecosystems, and the loss of sensitive plant species. The loss of vegetation cover can also reduce associated animal populations, as for example found with small mammals in the UK (Lake et al, 2001).

Although abandonment and under-grazing is a more extensive threat to most heathlands and moorlands, occasionally over-grazing impacts do occur; especially around areas where stock receive additional feed. Such concentrations of livestock can also lead to changes to plant species composition also results from a high density of dung that enriches the naturally poor soils. These impacts are likely to be relatively small-scale. However, of more concern are reports that large expanses of lichen pastures in Sweden and Finland (including both mountain heaths and mountain birch forests) are strongly influenced by reindeer grazing (Zaghi, 2008b). Reindeer herding forms one of the key pressures on tundra ecosystems in the EU Nordic countries (Finland and Sweden). As a widespread and dominant ungulate across many tundra and taiga regions, the reindeer plays an important role in affecting ecosystem structure and function (Forbes et al, 2006; Forbes and Kumpula, 2009). Animals, both free-ranging and herded, move seasonally between habitats and pastures. Their effects on vegetation and soils vary greatly in space and time depending on factors such as altitude/exposure, snow depth, substrate, moisture, prevailing vegetation type and, most importantly, animal density. Existing information indicates that degradation of pastures due to overgrazing is a serious problem for the Nordic tundra ecosystems (Kumpula et al, 2006; Raunio et al, 2008).

In conclusion, the area affected by overgrazing of most heathlands outside Scandinavia is likely to be very small, and probably less than 1%. More widespread overgrazing of tundra ecosystem occurs in Finland and Sweden, but the total area actually degraded is difficult to estimate (Raunio et al, 2008). Taking this into account, this study assumes that in total **2% – 5%** of heathlands and moorlands were affected by over-grazing in 2010.

The degradation level from over-grazing in 2020 is very difficult to estimate because recovery times will depend on the severity of the impact, the vegetation type involved and its grazing history. A further complication, is that some areas will be subject to annual grazing whilst others are only occasionally over-grazed, which may allow recovery. However, it is beyond the scope of this study to investigate further these complex issues. We therefore make a crude assumption that grazing impacts may typically last several years, and therefore the level of degradation from the cumulative impacts of over-grazing is **5% - 15%**. But it should be noted that, as further described in the next section, in most situations vegetation recovery should be possible through natural processes if grazing pressures are alleviated. Therefore proactive restoration measures will not be needed for most of the over-grazed habitat.

Inappropriate burning and accidental fires

Fire is a natural process in heathland ecosystems that can help maintain the habitat by preventing natural succession. Therefore intentional burning has been used as a common management measure to maintain (semi-) open landscapes like heathland, particularly in upland moorland in the UK. Burning also reduces litter and release nutrients, which stimulates earlier growth and temporarily increases the accessibility, palatability, and nutrient content of forage for livestock. Burning is also commonly carried out on upland heathlands in the UK to improve habitat conditions for Red Grouse (*Lagopus lagopus*), which are an important game species.

It is evident from a number of studies and reviews (Shaw et al, 1996; Tucker, 2003) that in appropriate areas and circumstances, carefully managed burning can play an important role in the maintenance of some open semi-natural upland habitats of high conservation importance. Fires may also help to maintain low nutrient conditions, which may be particularly important under current circumstances where atmospheric pollution is a significant pressure (see section 5.5.2). Burning of small patches can also increase vegetation structural diversity and species-richness in plants, invertebrates and birds of heathland habitats. Regular burning also reduces fuel loads and thus to some extent the risks of large and very hot wildfires.

However, inappropriate fires such as on peatlands or outside the burning season and inadequately managed fires have significant detrimental impacts, including soil and vegetation damage, and knock-on impacts on associated fauna. Furthermore, large wildfires sometimes develop from poorly managed fires, and their impacts can become extremely severe, especially when they occur during dry weather. According to recent conservation status assessments only some 800 ha of land within Sites of Special Scientific Interest (ie the main type of protected area in the UK) have an unfavourable conservation status as a result of intentional moorland burning[65]. Therefore, although there may also be significant areas outside the protected area network that may be more vulnerable to burning, and areas in Scotland, it is clear that the area of heathlands (and other habitats) affected by inappropriate burning are very small on a EU scale.

Of greater EU scale concern, is that wild fires and accidental fires also occur widely on lowland heathlands in drier areas and those close to urban areas. Typically some **10%** of lowland heathlands are at risk of fires that may cause long term damage as a result of their intensity and/or size. This is the area that should be subject to fire prevention measures (as further described below) although only a fraction of the area will probably burn in any one year.

The area of heathland and moorland that was actually degraded as a result of severe damaging fires in 2010 is uncertain, but it is likely to be a small percentage of the area at risk. Given the lack of data on the subject we use assume a broad range of 1% to 5% of the area at risk was detrimentally impacted by fires in 2010, and thus **0.1%** to **0.5%** of the total ecosystem area. Hence we assume that between 20% and 100% of the area at risk is likely to have been subject to damaging fires (eg very large or intense fires, or fires on fire-sensitive habitats) over the previous 20 years (ie the period over which the most significant fire impacts are likely to typically extend). On this basis between **2% and 10%** of the ecosystem is assumed to be have been degraded in 2010 as a result of the cumulative impacts of fires.

Drainage and low water tables

Reduced groundwater supplies may affect wet heathlands, making them less suitable for characteristic plants and affecting their buffering capacity to deal with nitrogen overloading. The combination of nitrogen availability and the hydrological situation determines the actual effects of these pressures on the habitats.

EU wide data on the hydrological impacts on heathlands and tundra ecosystems are not available. However, a proxy estimate can be deduced from the condition of wet heath Annex I habitat types (4010 and 4020), as monitoring data indicate that drainage and low water tables is a major cause of unfavourable status in these habitats. Together these habitats comprise 7% of the area of Annex I habitat types within this ecosystem type (see annex 1), and approximately 80% of their area has an unfavourable conservation status (ETC/BD, 2008a), which amounts to approximately 6% of the entire ecosystem type. Therefore, assuming between 10 - 50% of the unfavourable area is affected by hydrological pressures, and taking into account the plausible magnitude of error in this estimate this report assumes that **0.5-5%** of heathlands and tundra were degrade in 2010 as a result of the cumulative impacts of this pressure.

The rate of further drainage and reduced water tables occurring in 2010 is not known, but it is likely to have been low due to the currently low economic gains to be had from

65 http://www.sssi.naturalengland.org.uk/Special/sssi/reportAction.cfm?Report=sdrt17&Category=N&Reference=0 Accessed on 28/11/12

agricultural improvements in such ecosystems. We therefore assume only 1% - 2% of wet heaths is drained each year, amounting to **0.06% to 0.12% per year**.

Afforestation
Due to their poor soils and low agricultural profitability heathlands have tended to be converted to forestry rather than agriculture in recent years. Consequently significant areas have been converted to conifer plantations. Upland and wet heaths have mostly been planted with Spruce (Hampton, 2008). Dry heathlands in the Mediterranean have been planted with *Pinus pinaster* and *Eucalyptus globulus*. It is difficult to quantify the extent of afforestation on heathlands as heathland specific data are lacking. According to the Biodiversity Baseline Report about 67,000 ha of heathland and scrub ecosystem were lost between 2000 and 2006 (EEA, 2010a). Of this area 84% was lost due to afforestation. However, this assessment does not seem to include the large area of tundra in the EU, which is assumed to be only marginally impacted by afforestation areas. Therefore, after adding in the area of tundra to the assessment, and assuming afforestation rates in sclerophyllous habitats are comparable to heathland, afforestation affected only **0.01 – 0.02%** per year in 2010.

Conversion to agricultural land
Although afforestation is more common on heathlands and moorlands, some areas of lowland heath have been converted to grassland or arable land by burning, cultivation, liming, fertilisation, and reseeding. According to the Biodiversity Baseline Report about 5% of the 67,000 ha of heathland and scrub ecosystems were lost between 2000 and 2006 were converted to agriculture (EEA, 2010a). Therefore, as above, after adding in the area of tundra to the assessment, and assuming conversion rates in sclerophyllous habitats are comparable to heathland, agricultural conversion only affected less than **0.001%** of the ecosystem type per year in 2010. Furthermore, it is very likely that conversion of such habitats has declined in recent years to even lower levels as a result of protected area designations, CAP cross-compliance regulations and the reduced profitability of livestock farming especially on poor heathland soils.

6.6.3 Key pressures and extent of degradation in 2020

Specific pressures in 2020

Agricultural abandonment, undergrazing, afforestation and conversion to agriculture
It is considered that this ecosystem type will be affected by the same principle drivers of agricultural change as those affecting other marginal farming areas. As discussed in Box 6.1, recent modelling studies suggest that abandonment rates of 0.1% – 0.13% per year, appear most likely between 2010 and 2030. However, this is across all farmland and the magnitude of abandonment will undoubtedly vary considerably from place to place according to local circumstances. Therefore given the high vulnerability of moorland and heathland to abandonment and the likelihood that agri-environment spending levels will decline (see section 8.2) we assume here that abandonment rates will remain stable or increase slightly, such that in 2020 they will be **0.1% to 1.2%.** Consequently **4% to 41%** is assumed to be degraded in 2020 as a result of the cumulative impacts of abandonment.

As declining agricultural profitability and abandonment is a known driver of afforestation, afforestation rates are also expected to remain close to baseline levels, or even increase slightly, ie **0.01 – 0.05% per year**.

Due to expected improved regulations and reduced levels of profitability on marginal agricultural land, especially in countries with the majority of this ecosystem, it is expected that conversion rates to agriculture will decline to negligible levels.

Over-grazing
Over-grazing impacts are expected to decline to negligible levels by 2020, primarily as a result of better protection measures, guidance on environmental management, improved implementation and enforcement of CAP cross-compliance standards, and environmental and economic constraints on farming profitability.

Drainage and inappropriate burning and accidental fires
It is considered possible that drainage impacts could decline by 2020 due to better protection measures, guidance on environmental management and constraints on farming profitability, which make such capital investments increasingly uneconomic. Similarly, better protection and enforcement, management guidance and awareness raising may reduce the incidence of inappropriate management burning and arson. But these reductions in drivers of pressures may be offset by increasing impacts of climate change, which are already resulting in observable impacts on heathlands. It is not possible to quantify the net impact of these expected opposing drivers and therefore in the absence of better information this study assumes that these pressures will remain at baseline levels in 2020.

Overall degradation levels in 2020

Taking all the specific pressure trends into account, and respective foreseen measures to address them, this study assumes that as a reference scenario overall levels of degradation of heathlands will increase (primarily due to the risk of abandonment and under-grazing, and reduced agri-environment funding), to **20%** - **50%**.

Table 6-26 provides a summary of the estimated level of overall degradation and degradation resulting from each key pressure in 2007 and expected levels in 2020.

Table 6-26 Estimated baseline and expected 2020 percentage degradation rates, degradation levels and corresponding requirements to achieve a 15% restoration target, and conversion rates

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020		NB
	Min %/y	Max %/y	Min %/y	Max %/y	
Overall	**2.26%**	**6.62%**	**0.26%**	**1.82%**	
Agricultural abandonment and under-grazing	0.100%	1.00%	0.100%	1.20%	
Over-grazing	2.00%	5.00%	0.000%	0.000%	1
Inappropriate burning and wildfires	0.100%	0.500%	0.100%	0.500%	
Drainage and low water levels	0.060%	0.120%	0.060%	0.120%	

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels		Maximum level for 15% restoration[1]		Expected in 2020		% restoration required in 2020[2]	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Overall	10.0%	40.0%	8.5%	34.0%	20.0%	50.0%	11.5%	16.0%
Agricultural abandonment and under-grazing	3.00%	30.0%	2.55%	25.5%	4.00%	41.0%	1.45%	15.5%
Over-grazing	2.00%	5.00%	1.700%	4.25%	1.000%	2.00%	0.000%	0.000%
Inappropriate burning and wildfires	2.00%	10.0%	1.70%	8.50%	2.00%	10.0%	0.300%	1.50%
Drainage and low water levels	0.500%	5.00%	0.425%	4.25%	0.500%	5.00%	0.075%	0.750%

Notes: 1. Baseline degradation level minus 15%. 2.* % restoration required = expected degradation level minus the maximum level to achieve target.

c) Conversion rates

Conversion rates / cause	Conversion per year 2000-2010		Conversion per year 2010-2020		Total converted over 2000-2020		% re-creation required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Afforestation	0.010%	0.020%	0.010%	0.050%	0.200%	0.700%	0.030%	0.105%
Agriculture	0.000%	0.001%	0.000%	0.000%	0.000%	0.010%	0.000%	0.002%
Total	**0.010%**	**0.021%**	**0.010%**	**0.050%**	**0.200%**	**0.710%**	**0.030%**	**0.107%**

6.6.4 Key management and restoration measures

Heathlands are not climax ecosystems, but dynamic habitats that under natural conditions would often be transitory phases of natural succession unless this is checked by fire, grazing or other disturbances. Therefore in the absence of such natural conditions, heathlands generally require active management to arrest the process of succession to woodland, in order to maintain the structure and composition of dwarf shrub communities. Key maintenance techniques are **low intensity grazing management, mowing** and **scrub and tree management** (Hampton, 2008). Of these, carefully managed grazing by appropriate breeds of livestock is normally the preferred management technique, and should be carried out on most heathlands. In some areas, especially in upland heathlands in the UK **management burning** (ie controlled rotational burning of patches of dwarf shrub vegetation) is also necessary (Natural England, 2008). However, as inappropriate burning can also cause significant damage burning management needs to be carefully regulated and carried out according to burning plans. In some situations **grazing regulation** may be required to avoid, or enable recovery from, over-grazing impacts.

Restoration of degraded heathland may often be possible by re-establishing appropriate management measures, if the main problem has been neglect. In some situations management may need to be more intensive or modified for a period to deal with over-grown vegetation. For example, the removal of some scrub may be achievable by increasing grazing rates or the use of hardy livestock, such as some traditional cattle breeds, that are able to penetrate, break up and browse thick scrub.

In addition to, or to enable management, specific restoration measures to reverse the impacts of past pressures or to 'recreate' heathland habitats may also be necessary. These key restoration measures frequently include **tree and invasive species removal** (eg especially of non-native species that have planted, other invasive alien species, and undesired vegetation, such as bracken, that have developed as a result of inadequate or inappropriate management). In some cases **hydrological restoration** may be necessary, (eg by blocking drains and ditches, removing vegetation with high evapotranspiration rates such as deciduous trees, and creating hydrological separations between nature areas and agricultural landscape). Where nutrient levels have built up, or to help restore wet ground conditions, **turf stripping** by scraping away of the top soil may be required.

If heathland and moorland vegetation has been destroyed as a result of afforestation, agricultural conversion or extreme fires, then proactive **vegetation re-establishment**, may be required, especially if seed-banks have been lost or are no longer viable. In most situations natural recovery occurs from over-grazing when grazing pressures are reduced to appropriate levels, although this can be a slow process such that vegetation impacts can last for many years. If over-grazing has been severe enough to result in changes in vegetation composition then the ecosystem may remain in its acquired alternative stable state (see **Error! Reference source not found.**) and proactive vegetation re-establishment measures may be necessary. However, such measures are only likely to be needed over a few percent of the over grazed areas. Proactive measures are also needed where grazing has removed all vegetation such that erosion is preventing natural recovery. On moorlands this often happens on peat soils, but these impacts are addressed in section 6.8.

Other restoration measures may include the addition of alkaline material after turf-stripping to increase buffering capacity as mitigation for acidification, and establishing nitrogen capturing vegetation to reverse the past impacts of eutrophication. However, these techniques are expensive and/or used in limited situations and are therefore unlikely to significantly add to the estimates of restoration costs based on key measures. Source-targeted measures addressing on going air-borne pollution impacts that are not specifically related to moors and heathland alone are described in Chapter 5.

Additional important measures are often required to reduce ecosystem fragmentation, especially of lowland heathland habitats. However, as discussed in section 3.4.4 it is not possible to reliably estimate the costs of such measures.

Due to the extreme conditions that limit tree growth, tundra habitats are climax ecosystems and are therefore generally not dependent on active management for their maintenance, although **grazing regulation** may be required in some areas. However, **tree and alien species removal** and **hydrological restoration** may be required to restore areas of tundra that have been affected by afforestation.

Table 6-27 summarizes the key measures that address the key pressures on Europe's heathland and tundra, and provides an estimate of their percentage requirement with respect to each key pressure, based on the account above.

Table 6-27 Key maintenance and restoration and measures for heathland and tundra ecosystems and their required percentage coverage to address each key pressure

Degradation source / key pressures	Low-intensity managed grazing	Grazing regulation	Burning management	Burning plans	Mowing	Fire prevention and control	Scrub cutting and tree thinning	Turf stripping	Hydrological restoration	Removal of trees	Removal of invasive species	Vegetation restablishment
Agricultural abandonment and under-grazing	100		10		10		10			5	5	
Over-grazing		100										2
Inappropriate burning and wild fires				100		100					5	5
Drainage and low water table levels								5	95			
Afforestation										100		100
Agricultural conversion								60	10			100

6.6.5 Costs of key maintenance and restoration measures

An extensive literature exists on heathland management measures, but most articles focus on assessing the biological benefits or drawbacks of different management approaches. Although it is recognised that the spatial variability of costs of measures can be enormous (Naidoo et al, 2006), economic assessments of restoration projects are scarce in literature (Klimkowska et al, 2010a). Only a few published studies have actually included costs of measures when reporting on heathland management measures.

From an analysis of all the available cost data, the following conclusions can be drawn.

Combined maintenance costs

The cost estimates found by this study (from the UK, DK, NL and DE), for combined measures that aim to maintain heathland and tundra ecosystems range from 63 to 250 €/ha/y, averaging 164 €/ha/y. However, these cost estimates all relate to lowland heathland, and it is important to note that lowland heathland management costs tend to be significantly higher than for upland heathlands. This may in part be due to the large size of most upland heathland areas, because Strange et al (2007) reports that economies of scale may prevail in the costs of heathland management, as costs per hectare decrease with size of the area (from 78.5 €/ha/y for 100 ha to 26.6 €/ha/y for 7000 ha in Denmark). Using the lowland heathland costs would therefore be obviously biased. We have therefore calculated a combined upland heathland maintenance cost of 59 €/ha/y, from the specific costs of grazing and burning, as described below. Taking the average of these figures for lowland and upland heathland gives an average **combined maintenance management cost of 111 €/ha/y**. Although it might be argued that the average figure should take into account the larger area of upland heathland in the EU, it should also take into account the likelihood that per hectare costs may be much higher on small fragments of lowland heaths than indicated above. Therefore, because available data are inadequate to investigate these issues further, the simple average is used in this study, but it should be noted that it is subject to significant uncertainty.

It should also be noted that no data are available for maintenance costs for heathlands and tundra in Finland and Sweden, and heathlands in Spain, and therefore this may further contribute to uncertainty over the costs of combined management measures. Costs in Spain may be significantly lower than the average taken from primarily northern EU countries above. But on the other hand, as discussed above the Spanish heathland area may be over-estimated.

Costs of specific key maintenance measures

Low intensity grazing management costs estimates are low in the Netherlands (34-38 €/ha/y) and the UK (50 €/ha/y) and but are much higher in Sweden (120 – 270 €/ha/y) and Germany (150 to 220 €/ha/y). In Germany some scheme costs may are as high as 450 €/ha/y, but this probably reflects more ambitious measures (eg for nature reserves) and relatively small sites, especially compared to UK uplands. Therefore, the 450 €/ha/y estimate is considered atypical and is excluded from this cost calculation. On this basis the average of typical grazing costs are estimated to be **116 €/ha/y**.

Rotational burning is a relatively inexpensive measure and corresponds to the cost of the manpower (Hampton, 2008). In fact in the UK it is routinely carried out for livestock and game management purposes and additional payments for nature conservation is not therefore normally required. Nevertheless, it may be a requirement of some combined agri-environment scheme prescriptions in the UK, where supplementary burning requirements are compensated at 8.75 €/ha/y in England and 13.75 €/ha/y in Scotland. Muller (Muller, 2004) reports costs of 355 €/ha, but the measure may only be applied on 0.05/y for 2% of the surface. Using those figures, the data suggest a cost of 0.36 €/ha/y, while (Verheijen et al, 2009b) reports 0.57 and 1.14 €/ha/y. The rounded up average cost for burning is therefore assumed to be **5 €/ha/y**.

Fire prevention and control measures may firstly include the development of burning plans by landowners in consultation with environmental authorities for management burning of heathlands. These plans should be the responsibility of landowners and required to comply with regulations, and therefore no additional costs are likely to be involved. Secondly, land owners and authorities may provide warning signs and basic fire control equipment to help prevent accidental fires and to help with their control if they do occur. Data on the costs of these measures are not readily available but are likely to be relatively low, especially in northern Europe and when compared to Mediterranean areas with sclerophyllous vegetation. For this study it is assumed that such measures cost **1 €/ha/y**.

Mowing cost estimates show considerable variation ranging from 3 €/ha/y for wet heathland in the Netherlands to 400 €/ha/y for heathland in Germany. It is not possible to establish the reasons for these discrepancies without further detailed investigations. However, it does seem unlikely that the lowest cost will be typical given the practical difficulties of mowing many heathlands and the negligible value of the resulting cut vegetation. Therefore the low Netherlands estimate is treated as an atypical outlier, and the cost of mowing assumed to be **248 €/ha/y**, based on the average of four cost estimates (from DE, NL, SE and UK).

Costs of specific restoration actions

Tree felling, scrub removal and coppicing costs on heathland in the Netherlands vary between 41 and 44 €/ha on wet and dry heath respectively, ie averaging 42.5 €/ha. In contrast, costs in Germany have been estimated to be as much as 3,974 €/ha, but this is for dense scrub, so is not typical. For this study a rough estimate of **100 €/ha** is taken for scrub and tree removal.

The costs of **controlling alien and invasive species** other than trees, such as Bracken, will vary considerably depending on the species involved and the circumstances. However, the control of Bracken is a particularly common restoration action, and therefore the costs of controlling this species is taken as an indicative estimate of alien and invasive species control costs. According to UK data, Bracken control costs vary between 325 and 585 €/ha/y, averaging **455 €/ha**, which is used in this study as an indicative cost of controlling alien and invasive species.

No specific estimates of the cost of **hydrological restoration,** in the form of ditch blocking, within wet heathlands and tundra ecosystems appear to be available. Therefore, for the purposes of this study the costs of basic hydrological restoration of this ecosystem (ie the blocking of drains and ditches) are considered to be the same as those in mire ecosystems, as described in section 6.8. In mires ditch blocking costs are between 69 and 250 €/ha restored (Environment Agency, 2010; Verheijen et al, 2009b), averaging **169 €/ha**. It should be noted that the reliability of this estimate with respect to restoration costs in wet heathlands is uncertain.

In some **situations turf stripping** (and soil removal) is required to help reduce nutrients in the soil, and in some cases to help restore wet ground conditions. Cost data for these actions in the Netherlands, average 71 €/ha (69 €/ha on dry heath and 79 €/ha on wet heath). Data from Germany suggest that such costs can be as much as 3,100 €/ha, but this would be for certain sites requiring major restoration actions, and would not be typical of other areas that could be restored more cost-effectively to meet the 15% target. **71 €/ha** is therefore taken to be a reasonable estimate of the average cost of typical turf stripping restoration measures.

Vegetation re-establishment costs are only available for upland heathland in the UK, where agri-environment payment rates are **75 €/ha**. As there is little reason to believe these will differ considerably in other ecosystems and in the absence of other data, this cost estimate is taken to be indicative of typical costs for the ecosystem in general.

As with other grazed ecosystems it is assumed that **grazing regulation** (eg to limit stocking rates and grazing periods and to control the use of supplementary feeders) is most appropriately achieved through incorporation in GAEC requirement and/or national regulations. Agri-environment agreements also often include restrictions on potentially damaging activities, such as, for example, stocking rates or supplementary feeding. Therefore such actions are assumed not to incur additional direct costs.

Combined restoration costs

Agri-environment cost estimates and available studies of the costs of combined restoration measures show a considerable amount of variation, which probably reflects the ecosystem type being restored and its level of degradation and size, as well as Member States cost variability and the degree of restoration that is being aimed for. The lowest cost restoration appears to be through agri-environment schemes for upland heathland in England at 50 €/ha, with lowland heathland restoration in England being much more expensive at 250 €/ha. Combined restoration costs in the Netherlands are 287 €/ha and 380 €/ha in Sweden. The results of a LIFE project in France suggest restoration costs may be as high as 757 €/ha, but this was for experimental measures that may therefore be a typical. For this study, **345 €/ha** is taken as being the average of typical heathland and tundra combined restoration costs.

No cost data could be found on the specific re-recreation measures, but combined re-creation costs were estimated from 2012 agri-environment compensation rates in England. These costs vary greatly according to the ecosystem that is being replaced, with costs of all combined measures necessary for re-creating heathland in England being **250 €/ha** following forestry and **562 €/ha** following conversion to arable or improved grassland, averaging **406 €/ha**.

6.6.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration and re-creation targets. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). Two estimates are provided based on two methods: firstly the overall level of ecosystem degradation and average combined costs of maintenance and restoration (Table 6-28), and secondly, the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures (Table 6-29). See section 3.4 for a detailed account of the methodology and its assumptions.

The estimated average additional annual costs between 2010 and 2020 of **maintaining** the ecological condition for heathland and tundra ecosystems is between €2.58 to 18.1 million, according to overall degradation rates and combined measures costs, or between €2.25 to 17.1 million based on key pressures and the specific costs of key measures. There is little difference between the estimates from the two methods, but in both cases there are large differences between the minimum and maximum estimates, which reflects high uncertainty over degradation levels.

The estimated average additional annual costs between 2010 and 2020 of achieving the **restoration target** for heathland and tundra is between €36 to 50 million, according to overall degradation rates and combined measures costs, or between €199,000 to €1.9 million based on key pressures and the specific costs of key measures. This discrepancy between the estimates from the two methods is mainly due to the consideration in the key pressures and measures calculations of the proportion of the degraded area over which measures are likely to be required in practice. The low estimate takes into account the assumption that a large proportion of the degraded area would be able to recover through natural processes if basic maintenance management (eg grazing) was reinstated.

The estimated average additional annual costs between 2010 to 2020 of achieving the heathland and tundra **re-creation target** is between €109,000 to €387,000, according to overall degradation rates and combined measures costs, or between €67,000 to €242,000 based on key pressures and the specific costs of key measures. These relatively low costs reflect the relatively small area of the ecosystem that is expected to be converted between 2000 and 2020.

Table 6-28 Projected additional costs (€) in 2020 of achieving Target 2 in heathland and tundra ecosystems on the basis of the current costs of combined measures and expected overall degradation extent in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Ecosystem area = 8,949,182 ha

	Cost	Area required over		Cost (€) in 2020*	
Activity	**€/ha**	**Min**	**Max**	**Min**	**Max**
Maintenance required in 2020	111	0.26%	1.82%	2,580,000	18,100,000
Restoration required 2010-2020	345	11.50%	16.00%	35,500,000	49,400,000
Re-creation required 2010-2020	406	0.03%	0.11%	109,000	387,000

Note: * For restoration and re-creation the cost is averaged over 2010-2020

Table 6-29: Projected additional costs (€) in 2020 of achieving Target 2 within heathland and tundra ecosystems on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Duplicate costs are removed where "Included below" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 8,949,182 ha.

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

Key Pressure	% Area at risk in 2020		Key measure	Current annual costs (€/ha/y)	% area applied to	Instrument	Annual costs (€) in 2020	
	Min %/y	Max %/y					Min	Max
Agricultural abandonment and under-grazing	0.100%	1.20%	Low intensity grazing	116	100%	AEM	1,040,000	12,500,000
	0.100%	1.20%	Rotational burning	5	10%	AEM	4,470	53,700
	0.100%	1.20%	Mowing	248	10%	AEM	222,000	2,660,000
	0.100%	1.20%	Scrub cutting	100	10%	AEM	89,500	1,070,000
Over-grazing	0.000%	0.000%	Grazing regulation			GAEC	-	-
Inappropriate burning and wild fires	10.0%	10.0%	Burning management plans			GAEC	-	-
	10.0%	10.0%	Fire prevention & control	1	100%	AEM	895,000	895,000
							-	-
MAINTENANCE TOTAL							***2,250,000***	***17,100,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% requiring restoration in 2020		Key measure	Sub measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs (€) over 2010-2020	
	Min	Max						Min	Max
Agricultural abandonment and under-grazing	1.450%	15.500%	Tree and invasive species removal		100	5%	AEM	64,900	694,000
Over-grazing	0.000%	0.000%	Vegetation re-establishment		75	2%	AEM	-	-
Inappropriate burning and wild fires	0.300%	1.500%	Vegetation re-establishment		75	5%	AEM	10,000	50,300
	0.300%	1.500%	Tree and invasive species removal		100	5%	AEM	13,400	67,100
Drainage and low water table levels	0.075%	0.750%	Hydrological restoration	Drain blocking	169	95%	AEM	108,000	1,080,000
	0.075%	0.750%		Turf stripping	71	5%	AEM	2,380	23,800
RESTORATION TOTAL								***199,000***	***1,910,000***

ADDITIONAL RE-CREATION NEEDS TO REVERSE CONVERSION SINCE 2000

Key Pressure	% area requiring re-creation in 2020		Key measure	Sub measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs (€) over 2010-2020	
	Min	Max						Min	Max
Afforestation	0.030%	0.105%	Re-creation	From forest	250	100%	AEM	67,100	235,00
Agricultural	0.000%	0.002%	Re-creation	From agriculture	562	100%	AEM	-	7,540
RE-CREATION TOTAL								***67,100***	***242,000***

Notes: [1] It is assumed that additional drainage is not allowed on land in receipt of under agri-environment payments and is also prevented through appropriate regulatory measures elsewhere.

6.7 SCLEROPHYLLOUS VEGETATION

6.7.1 Description and extent

Sclerophyllous vegetation includes maquis, garrigue, mattoral and phrygana (see annex 1 for details). This vegetation cover is typical of the Mediterranean region, including Portugal (including the Canaries and Madeira), Spain, France, Greece, Italy, Malta, Cyprus, and Slovenia. It generally occurs on poor soils. The vegetation has developed under the Mediterranean climate with summer drought, and is the result of centuries of deforestation of *Quercus* forests for charcoal making, extensive grazing by sheep and goats, and regular fires. These habitat types are very dynamic and subject to rapid changes. For example, many of the dominant scrub species can grow into trees if left undisturbed, and sclerophyllous scrub is often found mixed with, and graduating into, various types of sclerophyllous broadleaved forest, such as *Quercus* spp., *Rhododendron ponticum* and *Laurus nobilis*. Many areas of sclerophyllous scrub habitat are also interspersed or scattered along coastlines, cliffs, and rocky areas. Scrub-grassland mosaics are particularly rich in biodiversity, for example invertebrates (Özden, 2009).

The extent of sclerophyllous scrub CLC in EU Member States is shown in Table 6-30, with the area of equivalent Habitats Directive Annex I habitat types for comparison. According to the CLC data most of the sclerophyllous scrub is in Spain, Italy, and France. According to the Habitats Directive habitat reporting, Spain has particularly large areas of Annex I thermo-Mediterranean / pre-desert scrub and juniper matorral (ETC/BD, 2008a), as does Italy, as well as the phryganas of *Genista* species on Sardinia. France has large areas of Annex I box scrub, mountainous *Cytisus purgans* scrub, and juniper matorral.

Acording to the CLC typology, the CLC sclerophyllous scrub area includes around 12,500 ha of *Juniperus communis* formations on heaths or calcareous grasslands in Sweden. The table also shows that a number of other Member States reported areas of Annex I juniper scrub habitat in association with heath or grassland but do not register any sclerophyllous scrub under the CLC system.

It is notable that the CLC data for sclerophyllous scrub is particularly unreliable as it is very difficult to distinguish scrub habitat from sclerophyllous forest, plus the habitat is very fragmented. The area for Greece is particularly low: Greece has over 14 times the surface area of Cyprus, and if it is assumed that it has a similar land cover, it ought to have at least 2.2 million ha of sclerophyllous scrub. There is also very little correspondence between the CLC area and the area of Annex I sclerophyllous scrub habitat reported under the Habitats Directive. Most notably, Malta and Slovenia reported vastly more Annex I habitat than was registered by CLC 2006.

Despite the limitations concerning the CLC data, they are used here to provide consistency with the other ecosystem types covered by this study. However, the CLC estimates for Greece, Malta and Slovenia are clearly too low to justify their use, so Annex I habitat areas are used for these countries, although they are still underestimates. Thus, the total area assumed in this study for sclerophyllous vegetation is **8,965,076 ha.**

Table 6-30 Area of CLC sclerophyllous vegetation (hectares), and equivalent Natura 2000 habitat types

The column shaded in grey denotes the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. The data for Greece are subject to large uncertainties. The transitional woodland-scrub area is regarded as including abandoned successional sclerophyllous scrub, and is divided up among four CLC land cover types according to their proportion in each Member State (see annex 1 for details). See annex 1 for details of Habitats Directive Annex I habitat types.

Land cover type	Annex I sclerophyllous scrub habitats in EU-25	CLC 323 Sclerophyllous vegetation	CLC Transitional woodland-scrub (share)	Total sclerophyllous scrub
Austria	30,400			
Belgium	140			
Bulgaria	n/a			
Cyprus	30,423	157,767	18,272	176,039
Czech Republic	1,304			
Denmark	1,000			
Estonia	5,300			
Finland	0			
France	113,630	572,314	38,403	610,717
Germany	6,145			
*Greece**	*340,408*	*877*	*183*	*340,408*
Hungary	1,500			
Ireland	156			
Italy	191,600	1,006,610	100,379	1,106,989
Latvia	60			
Lithuania	300			
Luxemburg	103			
Malta	18,700	4,982	0	18,700
Netherlands	400			
Poland	2,002			
Portugal	16,750	207,496	87,218	294,714
Romania	n/a			
Slovakia	2,372			
Slovenia	28,900	23	1	28,900
Spain	2,609,009	5,216,159	1,157,664	6,373,823
Sweden	8,600	12,526	2,260	14,786
United Kingdom	28			
Total	**3,409,230**	**7,178,754**	**959,363**	**8,965,076**

6.7.2 Key pressures and extent of degradation – 2010 baseline

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures in the baseline year, which for this study is taken to be 2010. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Ecosystem loss

The CLC land cover change analysis showed a loss of 268,200 ha of combined heath/tundra and sclerophyllous scrub area between 2000 and 2006, which is **2.9%** of the 2000 area (EEA, 2010a). Current habitat losses are due to development of infrastructure or urbanisation. Mountainous and coastal habitats are affected by the development of tourist infrastructure and human pressure, or to economic infrastructure development, transport networks and urbanisation. Mining and quarrying are a problem in some areas. There has been a historic habitat loss to the cultivation of olives, vines and grain. There is still habitat loss due to the clearing of areas for stock raising or agriculture, for example pre-desertic scrub (Natura 2000 habitat type 5330) is highly threatened by conversion to intensive irrigated agriculture (VV.AA, 2009). Afforestation with eucalyptus or pine plantations is causing loss of *Juniperus* scrub dune habitat (Picchi, 2008).

In the absence of more detailed information, this study assumes that between 2.6% and 3.1% (ie 2.9% +/- 10% plausible error) of sclerophyllous scrub has been lost between 2000 and 2006, which gives an annualised rounded loss of **0.4% to 0.6%.**

Overall degradation – baseline

The two key pressures on sclerophyllous scrub are the abandonment of extensive goat and sheep grazing and uncontrolled wildfires. However, sclerophyllous scrub is by its nature a very dynamic habitat, with considerable fluxes into forest or from regeneration of burnt areas or abandoned grassland (EEA, 2010a). Many of these recently vegetated areas will have a relatively low biodiversity. Because of the great range and variability of the habitats, and their dynamic nature, it is very difficult to delineate conservation objectives and target areas (VV.AA, 2009). The key pressures on the habitat and corresponding management measures are closely connected to the status of related habitats such as sclerophyllous forest, permanent crops, grassland or the broader coastline or mountain habitat. It is correspondingly very difficult to quantify these pressures (EEA, 2010a).

The uncertainty over the status of the ecosystem is evident from the Article 17 reporting on the status of Natura 2000 habitat types in 2007. Spain (where most of the habitats occur) reported an unknown conservation status for most of these habitats. Information from Italy, Greece and Cyprus was also either unknown or unfavourable. Mountainous juniper formations (habitat type 5130), which occur over a wide range of central and western Europe, were reported as being in unfavourable to bad conservation status. Of the total area of the ecosystem for which trends were reported 28% was in unfavourable condition. However, 72% was reported with unknown status, and no assessment data were provided from Romania or Bulgaria (ETC/BD, 2008a).

Taking this high level of uncertainty into account we assume that **between 18% and 38%** of sclerophyllous scrub is degraded in 2010 as a result of the cumulative impacts of pressures. The principal cause for further on going degradation in 2010 is likely to have been abandonment, as described below, and on the basis of this we assume the on going overall baseline rate of degradation was **0.2% to 1.0%.**

Specific key pressures – baseline

Abandonment of extensive goat and sheep grazing
Some areas of this type of vegetation in inaccessible places such as mountains or rocky coasts should be left alone, but most of the areas of sclerophyllous vegetation depend on regular disturbance through grazing plus some fire. Many of the secondary juniper matorrals are transitioning to forests through lack of grazing in Portugal, France and Italy (Calaciura and Spinelli, 2008b). Ungrazed areas are losing species diversity (Papanikolaou et al, 2011) and also become increasingly prone to damaging intense wildfires (due to the build up of high volumes of scrub, ie woody fuel). In southern Spain and Portugal, the habitat is linked to the traditional dehesa extensive grazing with cork oak, which is suffering from widespread abandonment.

Data on the specific rate of abandonment of sclerophyllous ecosystems are not available. However, it is likely to be at least in proportion to overall agricultural land abandonment rates as estimated in previous studies and discussed in section 6.2. But given their particularly hard climatic conditions and poor forage value, it is likely that abandonment rates may be higher than average. On this basis it seems likely that abandonment rates will be in the order of **0.2% to 1.0% per year**.

Furthermore, the effects of abandonment are exacerbated by the impacts of eutrophication. Shrubland communities typical of the Mediterranean region have not been well studied for their sensitivity to increased nitrogen deposition, but there are indications that their vegetation is sensitive to nitrogen deposition in combination with other factors such as drought or disturbance (Calvo et al 2005 quoted in (Dise, 2011)). See Chapter 4 for more details.

Uncontrolled wildfires
Sclerophyllous vegetation often depends on a certain level of fire for its continued existence as scrub rather than forest, but uncontrolled wildfire damage is a growing threat to this habitat, which is driven by abandonment of grazing, increasing human pressure, and climate change (see below).

Overgrazing
Overgrazing is a problem in some areas of the Mediterranean where large flocks of sheep and goats are still shepherded on common land, such as on Crete. Overgrazing and overburning are linked, as it is often shepherds who set off fires to try to produce more palatable vegetation for their animals (Papanastasis et al, 2002). Burned areas are heavily grazed because of their highly attractive new growth, and this causes a greater combined degradation than overgrazing alone. However, where shepherds are absent and there is also

no fire management system, the fire potential of the vegetation can build up and uncontrolled wildfires can damage large areas.

Impacts of invasive alien species and eutrophication are discussed in section 2.4.

6.7.3 Key pressures and extent of degradation in 2020

Ecosystem loss to 2020

Sclerophyllous scrub in mountainous and coastal habitats is likely to continue to be lost to the development of tourist infrastructure, urbanisation, and other development, due to the lack of recognition of the value of these habitats. Many scrub areas are on public lands managed by forestry organisations, who may increasingly recognise the value of scrub as a fire break, but who may also continue to use these lands for afforestation. In the absence of more detailed information, this study assumes loss of sclerophyllous scrub continuing at the same rate between 2010 and 2020, ie **0.4% to 0.6%.**

Key degradation pressures to 2020

Abandonment of extensive goat and sheep grazing
In some Mediterranean countries, such as Cyprus, free goat grazing of sclerophyllous scrub is considered to be ecologically damaging. However, in Portugal agri-environment payments are available for grazing management of maquis scrub (Keenleyside et al, 2012). Shepherds in Andalucia, Spain, receive agri-environment payments for the maintenance of fire breaks through grazing, which can maintain some intermediate successional stages of scrub, and this service can be expected to become increasingly important as wildfire risk increases (see below). However, current indications are that agri-environment funding may decline as a result of the reduced EU MFF and proposed reforms to the 2014-2020 CAP. This together with declining profitability of extensive livestock systems and the added impacts of climate change are likely to further increase the rate of abandonment. However, the actual magnitude of the likely increase is too uncertain to estimate.

Inappropriate burning and wildfires
An important concern is that climate change will result in more widespread and severe fire damage that will severely damage the biodiversity of the remaining habitat. On average, the annual area burned by wildfire might increase by a factor of 2 in Southern Europe, by a factor of 5 in Central Europe and by a factor of 3.5 in northern Europe as a result of climate change[66].

Eutrophication from nitrogen deposition
A business-as-usual scenario estimates that by 2050 approximately 69% of the Mediterranean region will have nitrogen deposition exceeding 10 kg N ha^{-1} yr^{-1}. This is widely considered as a threshold for nitrogen impacts (Bobbink et al 2003 and Phoenix et al 2006 quoted in (Dise, 2011).

[66] http://www.ecochange-project.eu/newsletter-and-briefing-sheets/stay-informed-2013-get-involved/ecochange-newsletter-n-5

In conclusion, there is sufficient evidence that pressures on abandonment, fire and eutrophication are significant, and likely to continue to 2020. However, available information is not sufficiently complete to estimate their likely extent, even in very broad terms. Therefore this study does not attempt to calculate costs of restoration on the basis of key pressures and related key measures.

Overall degradation level

Given the uncertainty over the changes in specific pressures it is very difficult to reliably estimate the overall degradation level in 2020. Therefore for the purposes of this study and in the absence of better data we assume the minimum level remains the same as 2010 (ie **18%**), but taking into account the likely increases in pressures described above, we assume the maximum level of degradation may increase to as much as **50%**. Similarly the rate of degradation may be between **0.8% and 2%** per year in 2020.

6.7.4 Key management and restoration measures

Maintenance measures

Management of sclerophyllous vegetation almost always involves extensive grazing, often with shepherding, which needs to be adjusted to the conservation targets for that habitat type and also adjusted as the habitat status improves. This grazing will only be viable if it is linked to the sympathetic management of related grazed grassland habitats, such as dehesas. Management also requires measures for wildfire control and prevention, including the maintenance of fire breaks (eg by grazing), and perhaps some managed rotational burning in particularly high risk areas.

Restoration measures

Restoration of degraded sclerophyllous vegetation requires the management measures above plus the removal of trees and invasive species such as *Eucalyptus globulus* and *Opuntia.* Protection from visitor pressure through fencing, paths and information provision is required in heavily frequented sites. The regulation of hunting may be necessary in some habitats .Many restoration projects focus on regeneration of the habitat after large-scale wildfire damage[67]. Soil erosion after fires is a big problem[68]. The protection of soil and vegetation regeneration after wildfires can be carried out through application of an organic layer of mulch, either alone or combined with seeding native grasses, or log dams or contour-felled barriers. Grazing can be controlled on fire regeneration sites by fencing or removal of animals (Ramón Vallejo et al, 2012). Regeneration of severely degraded scrub habitats involves the replanting or sowing of typical plant species, including collection or propagation of local genetic material, preparation / treatment, and dispersal or planting. Some projects have used hydro-seeding, which is a technique that mixes seeds, fertilizing and setting substances, water and a 'mulch' material.

[67] http://www.env.aegean.gr/labs/Remote_sensing/publications/full%20paper_Christakopoulos%20et%20al_Earsel.pdf

[68] http://uaeco.biol.uoa.gr/files/PDF/FP0701/Ch_05.pdf

6.7.5 Costs of restoration and management measures

It was difficult to source restoration and management costs for these vegetation types. Data on habitat restoration costs available for this habitat type are only relative to a few LIFE projects for which it was possible to access the relevant technical and financial information. It was not possible to carry out the detailed costs calculation for specific key measures, so only costs data for combined measures have been used here.

Combined maintenance costs:
In Portugal, agri-environment payments are offered for grazing and mowing of maquis glades (mowing maximum once a year, minimum every two years, removal of cuttings, restricted mowing dates; and/or grazing at maximum 0.8 LU / ha; scrub maintained on 10-50% of area) (Keenleyside et al, 2012). This is supported with 200 €/ha/yr. A similar agri-environment scheme for restoration mowing and grazing is supported with 450 €/ha/yr.

*This study assumes a combined maintenance cost for sclerophyllous scrub of **200 €/ha/yr**.*

Combined restoration costs:
Some indications of the cost of sclerophyllous scrub restoration can be deduced from the following projects, which were carried out mainly in protected areas:

- Restoration of 28 ha of arborescent juniper matorral through the removal of *Pinus halepensis* cost **2,429 €/ha** (mainly for the complete clearing of 3 ha, including felling of 5,000 plants, girdling of 1,500 plants, removal of 600 mt of wood) on Pianosa island in the Tuscan archipelago, Italy (LIFE04 NAT/IT/000172).
- Restoration of 120 ha of scrubland destroyed by wildfire along steep slopes using innovative hydro-seeding techniques cost **6,988 €/ha** (including preliminary studies; area reclamation; vegetal material collection for seeds and rhizomes, treatment and preparation of material for hydro-seeding; monitoring of results) in Cilento National Park, Italy (LIFE98 ENV/IT/000171).
- Arborescent juniper matorral (and heathland) restoration cost **605 €/ha**, including a management plan and a series of tests of experimental measures (including burning, manual or mechanical crushing, pasturing by sheep, cattle and horses) in Montselgues, Southern France (LIFE05 NAT/F/000135).
- Replanting juniper woodland and moorland damaged by erosion cost **3,377 €/ha** (including plant acquisition, clearing, hydro-seeding and planting of 200 plants[69] and complementary care, demarcation of paths and information boards) in the National Park Duratón River Canyon (Castilla y León), Spain (LIFE04 NAT/ES/000036).
- Restoration of over 50 ha of Macaronesian juniper woods cost **2,428 €/ha**, including removal of alien plants and planting of almost 15,000 native plants, on the Canary islands (LIFE04 NAT/ES/000064). The production of the planting material of endemic *Juniperus turbinata* ssp. *canariensis* cost 1.35 €/plant (400 €/ha).

*This study uses the average per ha cost of these projects to derive an average restoration cost for sclerophyllous scrub of **2,000 €/ha**.*

[69] 100 *Rosa canina*, 60 *Juniperus thurifera*, 60 *Juniperus oxycedrus* and 20 *Crataegus monogyna*

Habitat re-creation costs:
In the absence of specific costs data, it is also assumed that the re-creation of sclerophyllous scrub from sclerophyllous forest will cost **2,000 €/ha**.

6.7.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration and re-creation targets. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). Due to inadequate available information on the extent of key pressures and costs of key measures only estimates of combined maintenance and restoration costs in relation to overall degradation are calculated. See section 3.4 for a detailed account of the methodology and its assumptions.

The results provided in Table 6-31 indicate that the estimated additional annual cost of **maintaining** the ecological condition of sclerophyllous scrub is between €14 and 36 million. The annual additional cost averaged over the 2010 to 2020 period of achieving the **restoration target** is between €48 and 317 million, whilst the averaged additional annual costs of meeting the **re-creation target** is between €21.5 to 32 million.

It should be borne in mind that the data for sclerophyllous scrub are characterized by considerable uncertainties and the area of the ecosystem is probably significantly underestimated, especially in Greece. This is combined with a lack of consensus in the conservation community about how to manage these habitats. It is, however, clear that in the future this habitat will need to be managed more intensively for fire control and prevention, including extensive grazing to reduce succession and litter build up, and grazing of fire breaks. The habitat will also require integrated management measures that encompass also the grassland (eg dehesa) and woodland habitats to which sclerophyllous scrub is connected. For all of these reasons the cost estimates for this ecosystem should be treated with particular caution.

Table 6-31 Projected additional costs (€) of achieving Target 2 in sclerophyllous scrub on the basis of the current costs of combined measures and expected overall degradation extent in 2020

Important notes: All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. See section 3.5 for further explanation.

Ecosystem area = 8,965,076 ha

Activity	Cost €/ha	Area required over		Cost in 2020*	
		Min	Max	Min	Max
Maintenance required in 2020	200	0.80%	2.00%	14,300,000	35,900,000
Restoration required 2010-2020	2,000	2.70%	17.70%	48,400,000	317,000,000
Re-creation required 2010-2020	2,000	1.20%	1.80%	21,500,000	32,300,000

Note: * For restoration and re-creation the cost is averaged over 2010-2020.

6.8 MIRES (BOGS AND FENS)

6.8.1 Description

Mires are permanently waterlogged habitats that have peat-accumulating vegetation specially adapted to the anaerobic conditions. Mires have accumulated their peat over very long periods of time, with an average of 1mm of peat accumulation a year in active systems. They store a very significant amount of carbon. The CLC vegetation class 412 includes all bogs and fens (collectively known as mires) characterised by plant communities associated with a thick peat layer (more than 30cm of peat) and less than 30% tree cover (see annex 1 for more details on these habitats) (Büttner et al, 2006). As it is impossible to verify depth of peat from satellite imaging, the CLC mire distribution is in effect therefore determined by the presence of the characteristic vegetation types.

Mires include rainwater-fed highly oligotrophic raised bogs, groundwater-fed oligotrophic to mesotrophic fens, and a range of other types. A mire landscape can display a complex combination of habitat types; upland blanket bogs are often interspersed with nutrient poor fens (trackways and ladder fens), and raised bogs can grade into fringing 'lagg' fens. Some mires are covered in trees, such as bog woodland, spruce mires, or ash, black alder and birch forests; these areas are registered in the CLC classes for woodland. For example, more than half of the total mire area in the Nordic countries is wooded (Normander et al, 2009). Mires are also very difficult to separate from wet heath and scrub habitats, into which they grade, and which they convert into as a result of overgrazing. Fen meadows, which are permanently waterlogged below the soil surface, are discussed as semi-natural grassland in section 0. Some rich-fen habitats, including sedge and reed mires, with taller vegetation and a thinner peat layer, are included in the inland marshes section 0. Small areas of transitional or quaking mire along the margins of lakes will also be included as lake or inland marsh in section 0.

Annex I of the Habitats Directive lists 9 mire habitat types, including active raised bogs and blanket bogs, transition mires, alkaline fens, aapa mires and palsa mires, plus bog woodland and dystrophic (peatland) lakes (see annex 1 for details). Degraded raised bogs may also be registered by this CLC class if the peat layer is still thick enough, but if the peat has been too damaged through drainage, burning, or erosion, then the habitat will become *Molinia* grassland or heath (JNCC, 2011), or become invaded by bracken, scrub such as gorse, and eventually woodland. If the area is heavily grazed after drainage, it often becomes acid *Nardus stricta* grassland.

6.8.2 Extent of mires

A summary of the approximate areas of bog and fen land cover in each Member State is provided in Table 6-32.

Table 6-32 Areas of mire CLC (hectares), equivalent Natura 2000 Annex I habitat areas, and area of peatland assessed by the International Mire Conservation Group in 2008 for comparison

The column shaded in grey denotes the area data that were used in this study. CLC data relate to 2006 CLC data except where indicated as * where data are for 2000. The data for Greece are subject to large uncertainties. Annex I habitat areas accord to data submitted to the European Topic Centre on Biodiversity in 2007 and do not include data for Romania or Bulgaria. Areas of peat and peat-topped soils by country are estimated from the Map of Organic Carbon in Topsoils of Europe (and the associated 1km data set), using threshold of 25% OC (Montanarella et al, 2006). The area of peatland reported by Member States in 2008 and proportion of degrading peatland is taken from the IMCG inventory (Joosten, 2009). Where the proportion is over 100%, the total degrading peatland area was reported as larger than the peatland area.

Land cover (ha)	CLC 412 Peat bogs and fens	Equivalent Natura 2000 habitats (EU-25)	Peat soil with at least 25% organic matter	Area of peatland reported in 2008 (IMCG)	% of peatland reported as degrading in 2008 (IMCG)
Austria	2,093	29,000	13,400	18,900	63%
Belgium	5,034	2,417	9,500	14,600	110%
Bulgaria	1,263	n/a	0	11,200	63%
Cyprus		0	n/a	100	0%
Czech Republic	4,171	6,836	25,100	25,000	88%
Denmark	22,893	16,560	6,600	127,600	108%
Estonia	123,815	268,800	688,900	943,000	52%
Finland	2,200,055	2,593,313	9,835,300	7,942,900	80%
France	4,616	855,005	77,500	140,000	80%
Germany	89,088	78,949	627,900	1,666,800	78%
*Greece**		*157*	0	6,600	82%
Hungary	9,607	2,310	40,100	30,100	96%
Ireland	1,087,212	450,313	1,272,500	1,109,000	34%
Italy	364	17,600	100	19,100	52%
Latvia	134,346	245,833	338,200	639,000	31%
Lithuania	39,342	44,300	148,900	327,900	84%
Luxemburg		21	0	270	74%
Malta		0	0	-	-
Netherlands	7,801	6,921	202,200	345,100	67%
Poland	8,480	19,770	467,700	1,152,800	88%
Portugal		37,416	0	1,860	81%
Romania	833	n/a	3,900	96,200	44%
Slovakia	193	1,617	100	11,800	93%
Slovenia		565	0	7,400	91%
Spain	579	3,892	18,400	5,700	65%
Sweden	2,825,948	2,909,157	9,078,500	6,562,300	20%
United Kingdom	1,107,572	2,231,181	4,441,100	1,711,300	25%
TOTAL	**7,675,305**	**9,823,933**	**27,295,900**	**22,916,530**	

The CLC land cover shows 7.67 million ha of mire in 2006. In 2007, Member States in the EU-25 reported over 9.8 million ha of the equivalent Natura 2000 habitat types (11% raised bogs, 47% blanket bog & transition mires, 11% fens), plus 2 million ha of bog woodland (see annex 1 for details). The CLC data for mire areas is known to be inaccurate (Martinez Sanchez et al, 2011), and there are some striking discrepancies between the Annex I habitat data reported by Member States and the CLC area. Mire data were gathered by the International Mire Conservation Group in order to calculate peat-carbon stocks and emissions for international climate change negotiations (Joosten, 2009), and these areas are three to ten times greater than the CLC data. As mire habitat is heavily concentrated in a few countries, it is also possible to use other data sources for these countries (see annex 1 for details). The comparisons show that the CLC data is significantly underestimating the extent of mires in Sweden, Finland, Latvia, Estonia, and Lithuania, probably because of the presence of trees, and also underestimating their extent in Poland, the UK and France, probably because of degradation to grassland or scrub. In contrast, the data for Ireland appears to be reasonably accurate; however Ireland is only reporting a fraction of this area as Annex I habitat. It should also be remembered that Central European countries still have many excellent examples of peatland types that are virtually extinct further west (Bragg et al, 2003).

Some indication of the potential for restoration of peatlands in the EU is given by the distribution of peat soils, as shown in Table 6-32 (Montanarella et al, 2006). However, a lot of peat soils in northern and western Europe are the remains of prehistoric or historic peat accumulation under climatic or local hydrological conditions that no longer exist. Nevertheless, as the CLC data relies on the presence of characteristic bog or fen vegetation, and does not include degraded peatland that is covered in bracken, grass or heath, the CLC data is likely to be a significant underestimate of the amount of area that could theoretically be restored to bog or fen habitat.

This study uses the CLC area data for bogs and fens of 7,675,305 ha (whilst recognising that this is likely to be an underestimate).

6.8.3 Key pressures on mires and extent of degradation – 2010 baseline

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures at the baseline year, which for this study is taken to be 2010. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Ecosystem loss

A vast share of Europe’s peat bogs and fens have already been destroyed and cannot be restored because the peat soil is too degraded. The Netherlands, for example, have destroyed almost all of their natural peatlands (Pakalne, 2004). The rate of loss has slowed in the last decade; between 2000 and 2006 the CLC Land Cover change analysis[70] recorded a loss of 21,500 ha of inland marshes (CLC411) and mires (CLC412) together (whereas since

[70] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

1990 126,700 ha have been lost) (EEA, 2010a). Historically, large areas of peatland have been drained for conversion to agricultural use. Drained peatland may be cultivated or harrowed, and then treated with lime and fertiliser and seeded to produce improved grassland, or used to cultivate cereals, field vegetables, or root crops. This leaves the peat surface bare for periods of the year, which increases mineralisation and erosion of the peat soil. Peat soil from fens or bogs that has been turned into arable land generally has very limited potential for restoration, and is mostly classified as wasted peat (see below). In the **UK**, 40% of peatlands have been modified or destroyed by conversion to agriculture (Littlewood et al 2010 quoted in (Bain et al, 2011). Peat bogs and mires continue to be lost to infrastructure developments or mining. For example, in **Estonia**, some 2,000 ha of mire have been destroyed by opencast mining of the oil shale beneath, and this activity advances across another 100 ha of mire each year (Minayeva et al, 2009).

Assuming the CLC area loss applies proportionally to inland marshes and mires according to relative area, it is assumed that at least 14,629 ha of mire was lost from 2000 to the baseline, ie 0.9% of the 2000 area, equivalent to an annual loss of 0.09%. Taking into account the likely error range in this estimate, we assume in this study that the annual rate of loss from 2000 to 2010 was **0.08% - 0.1%.**

Overall degradation – baseline

The principal cause of degradation and loss of peat bogs and fens is drainage and hydrological modification, which destroys the peat-forming vegetation (EEA, 2010a; Šefferová et al, 2008). It is likely that peat formation has stopped on over 60% of Europe's original mire area (Joosten, 2009). Bogs and fens are also very sensitive to changes in hydrology and hydrochemistry in the wider area including the whole catchment, so are also damaged by drainage and reclamation of surrounding fields. Blanket bogs depend to a large degree on the maintenance of surface water flow patterns at a landscape scale and hence are dependent on sensitive land management practices. The overall conservation status for alkaline fen habitat is bad (Šefferová et al, 2008). Fens have experienced a decline in quality as a result of peat mining, draining for cropland, infilling, and fertiliser pollution and eutrophication. In Ireland, for example, only limited measures have been introduced to address these damaging activities, which are likely to have increased in severity since the 1990s (National Parks and Wildlife Service, 2008). Similarly, the status of transition mires is bad because of drainage, infilling, land reclamation and pollution.

According to the Article 17 reporting on Annex I habitats, 92% of the area of bog and fen habitats in the EU was in unfavourable condition in 2007. The monitoring of these Annex I habitat types is generally good with no major gaps in coverage (ETC/BD, 2008b). As the Annex I habitat locations should correspond well with the CLC locations, and the Article 17 assessments in general cover a much larger area than the CLC coverage (Table 6-32), it can be assumed that the Article 17 assessments provide a reasonably representative and reliable indication of the level of degradation of the ecosystem type[71]. An assessment by the International Mire Conservation Group in 2008 found most Member States reporting 65% to

[71] However it should be noted that Romania and Bulgaria did not report on their bog and fen in the Article 17 report, which could be 2-5,000 ha, and is likely to be predominantly degraded.

100% degradation of peatland habitat (see Table 6-32) (Joosten, 2009). As 93% of the CLC mires area is found in Sweden (37%), Finland (29%), the UK (14.5%) and Ireland (14%), the status in these Member States determines the overall degradation level of the ecosystem.

Sweden reported a degradation level of 17-22% of the area of its Annex I bog and fen habitat in its Article 17 report, and 20% of degraded mire to the IMCG in 2008. A different reference assumes that 47% of Sweden's mire area is still intact (Vasander et al, 2003).

Finland assessed 18-27% of its Annex I habitat area as degraded in its Article 17 report, but reported 80% of degraded mire for a much larger area to the IMCG in 2008. A different reference assumes that 40% of Finland's mire area is still intact (Vasander et al, 2003).

The UK reported 25% degradation to the IMCG, but a recent report estimated that more than 80% of the UK's mires are damaged and the majority are no longer forming peat (Littlewood et al 2010 quoted in (Bain et al, 2011). England's estimates of the state of its peatland (Bain et al, 2011; Natural England, 2010) are that only 1% of deep peat is mapped as undamaged, waterlogged and actively sequestering carbon (active peat); 30% is drained and supports non-peat forming semi-natural vegetation – either grass or heath (degraded peat)[72]; 24% of deep peat is under cultivation (archaic peat); and the rest is under species-poor improved grassland or other land uses.

Ireland only reported under Article 17 on an equivalent of around 40% of the CLC mire area, and it can be assumed that this difference in area is because most of the habitat is too degraded to be considered to be Annex I habitat. Intact active raised bog has decreased in area by over 35% in the last 10 years and is now a very rare habitat; maybe a third of Ireland's remaining peat habitat is degraded raised bog still capable of regeneration, but the future prospects of this area is poor unless active restoration measures are undertaken soon (National Parks and Wildlife Service, 2008). Extensive areas of blanket bog have been removed or highly modified (chiefly through drainage, reclamation, peat extraction, afforestation but also via erosion and landslides triggered by human activities). Current pressures include overstocking, peat extraction, drainage, burning and infrastructural developments.

Taking these factors into account, including that some mires are not Annex 1 habitats, we consider that **80 – 95%** of the ecosystem area is degraded.

Specific pressures – baseline

Disturbed hydrology and drainage

As the principal cause of degradation of bogs and fens is disturbed hydrology, this is the most quantified pressure on the mires that remain. In a recent nation-wide inventory in Sweden, disturbed hydrology was reported on two-thirds of the almost 1,600 alkaline fen sites with high nature conservation values (Šefferová et al, 2008). A different study assumes a drainage rate of 24% of Sweden's mires (9.6% for agriculture and 14.4% for forestry)

[72] this can be valuable habitat and the carbon store may be maintained although peat is no longer forming, but some areas are dominated by species-poor bracken or acid grassland, or is overgrazed or overburnt

(Vasander et al, 2003). Finland's Peatland Strategy 2011 reports that 45% of its mire area is intact and 52% is drained[73] (MMM, 2011). Other national data states that 55% of Finland's 16,600,000 ha of mires is degraded by drainage, mainly for afforestation (Normander et al, 2009). A different study assumes a drainage rate of 61% of Finland's mires (6% for agriculture and 55% for forestry) (Vasander et al, 2003). Over the last 50 years, the area of pristine (undrained) mire habitat in Finland has decreased by almost half, whilst Finland's populations of specialist mire butterflies have decreased by almost 50% in the last twenty years (Normander et al, 2009). Only 12% of the UK's mires are found in England, but estimates of the state of its peatland (Bain et al, 2011; Natural England, 2010) are that 30% is drained and supports non-peat forming semi-natural vegetation (degraded peat)[74]. In Ireland, on-going deterioration of the hydrological conditions of bogs caused by peat cutting, drainage, forestry and burning severely threatens the viability of the habitat at most locations (National Parks and Wildlife Service, 2008).

This study assumes that in 2010 **40-50%** of mires were degraded as a result of the cumulative impacts of drainage and disturbed hydrology. The annual rate of degradation from this pressure in 2010 is unknown, but it is likely to have been declining.

Afforestation
Afforested peatland is normally heavily altered by forestry operations such as soil preparation, thinning, and clear cutting, as well as by the necessary drainage, but some original bog vegetation may survive in open areas (JNCC, 2011; Normander et al, 2009). Afforestation contributes further to the drying out of peatland through the action of the tree roots that withdraw water and create pores that aerate the soil and further stimulate decomposition. Drainage is sometimes carried out several times; this is the case on more than 2 million ha in Finland (Normander et al, 2009). Some incompletely drained or recently afforested peat can be classified as archaic peat still capable of some degree of restoration, but restoration of peatland after afforestation is not always successful (Anderson, 2010).

In Sweden, around 15% of peatlands are affected by forestry (Vasander et al, 2003). In Finland, over half of the mire area is degraded by drainage for afforestation projects (Normander et al, 2009; Vasander et al, 2003). In the UK, there has been a steep decline in new afforestation on deep peat soils since 1990 (Bain et al, 2011), but overall, 10% of the area has been afforested with conifers (Littlewood et al 2010 quoted in (Bain et al, 2011). In Ireland, the area of formerly intact mire afforested with conifers is estimated at 216,400 ha, which is equivalent to 20% of the CLC 2006 area (Farrell, 2008).

This study assumes that in 2010 **15-25%** of mires were in a degraded condition due to afforestation. The annual rate of degradation from this pressure in 2010 is unknown, but it is likely to have been declining due to improved protection.

[73] Finland reports that 47% of its mire area is intact and 53% is drained. If this area is combined with the reported area of active peat extraction and area of grassland on degraded peatland, then the proportion intact is 45%, the proportion drained is 52%, the proportion under active peat extraction is 3% and the proportion of degraded peat under grassland vegetation is 1% (MMM, 2011).
[74] this can be valuable habitat and the carbon store may be maintained although peat is no longer forming, but some areas are dominated by species-poor bracken or acid grassland, or is overgrazed or overburnt

Peat extraction

Peat cutting in raised bogs and blanket bogs continues to be a significant threat to these habitats in Ireland (National Parks and Wildlife Service, 2008), Finland, Estonia[75], and Latvia (Minayeva et al, 2009). In Sweden, peat harvesting is now performed on around 0.1% of the original mire area, for both energy and horticultural purposes (Vasander et al, 2003). A study estimates the area subject to active peat cutting or past peat cutaway areas at around 1% of Finland's mire area (Vasander et al, 2003). In the UK, 11% of peatlands are affected by past peat cutting (Littlewood et al 2010 quoted in (Bain et al, 2011). The area of raised peat bog actively exploited for peat cutting in Ireland is estimated at 100,000 ha, 9% of the CLC 2006 mire area and 7.5% of Ireland's original peatland area (Farrell, 2008). A recent Irish NGO report assessed 33 bogs and found damage that will result in a further deterioration of their conservation status - cutting, burning, draining, and adjacent extraction - at 22 of them (Friends of the Irish Environment, 2011).

Industrial peat extraction results in a complete loss of the pre-existing peatland ecosystem character. Once the commercial peat has been removed, the production fields have variable peat depth (0.5m to >2m); remaining peat is compressed and there are large scale fluctuations in the water-table (Farrell, 2008). The potential to restore these areas to lowland raised bogs is minimal, and can only be done where peat remains above the influence of alkaline waters. However, most cut bogs are then further utilised for energy peat, which generally brings the surface of the cutaway bog down to a level of influence from alkaline groundwater. Abandoned peat cuttings are then often flooded, resulting in areas of open water. These can accumulate sediment and develop a fringing or floating peat-forming fen vegetation, but the water is not shallow, still, acidic and nutrient poor enough to allow bog habitats to reform (JNCC, 2011). Sympathetic restoration can create a mosaic of semi-natural wetland habitats and acid grassland, heathland and birch woodland, but not restoration of active raised bog.

This study assumes that **2-10%** of mires were in a degraded condition in 2010 due to peat extraction and/or past peat cutting. The annual rate of degradation from peat extraction and peat cutting in 2010 is unknown, but it is likely to have been declining due to improved protection and possibly declining demand.

Lack of maintenance grazing and mowing

Most intact core bog and fen habitats are best left without any form of management. However, around half of the EU's mire area consists of blanket bog, transition mire or fen (see annex 1), and most of these habitat areas will need some low intensity management to prevent succession and to counter the effects of eutrophication (Halada et al, 2011). Much of the mire area affected by drainage will also need some ongoing management in the form of scrub clearance, even on the areas where the hydrology is restored (see below for details). Traditional management practices of extensive mowing (hand mowing, late mowing, no fertilisation, every 1-2 years) and light grazing have been abandoned in many marginal areas, eg in Biebrza National park, Poland where 20,000 ha of abandoned fens need restoring (Šefferová et al, 2008).

[75] A study estimates that 3.5% of Estonia's mire area is affected by current or past peat cutting (Vasander et al, 2003).

Only 12% of the UK's mires are found in England, but it is estimated that as a result of drying and inadequate grazing 4% is covered in scrub and 3% in woodland (degraded peat to archaic peat) – some is valuable habitat but other areas are dominated by invasive *Rhododendron ponticum* (Bain et al, 2011; Natural England, 2010).

Overall, this study assumes that **40-50%** of the mire ecosystem area is impacted by the effects of inadequate maintenance management by grazing and/or mowing, with some occasional scrub clearance. The annual rate of further abandonment /under-management in 2010 is difficult to assess as ecosystem specific data are not available. But it is likely to be have been at least in proportion to overall agricultural land abandonment rates as estimated in previous studies and discussed in section 6.2. But given their particularly poor forage value and difficult farming conditions, it is likely that abandonment rates may be higher than average. On this basis it seems likely that abandonment rates will be in the order of **0.2% to 1.0% per year**.

Overgrazing

Mire habitats tend to be very sensitive to grazing and trampling by livestock. In Ireland extensive areas of blanket bog have been damaged by overstocking, and some damage continues (National Parks and Wildlife Service, 2008). In the period 1987-1997 it is estimated that overgrazing damage had left 7% of Ireland's total blanket bog area severely damaged with a further 7% under the same threat (Foss 1998 in (Farrell, 2008)).

However, much of the past overgrazing was a result of CAP headage payments that were linked to the numbers of livestock owned by a farmer, and which therefore encouraged high stocking rates. Now CAP payments are almost entirely decoupled from livestock numbers and over-grazing of semi-natural habitats has declined markedly. Therefore, although current over-grazing levels are uncertain at an EU scale they are likely to be very low.

Burning and wildfires

Burning has been used as a management tool to create grazing land on drained peatland, or to clear the peat soil for cultivation. For example, burning has been used extensively in Romania to create arable land on peatlands along the Danube River. Burning has been used in the UK on blanket bog to manage heather regrowth for grouse, and to create foraging for deer and livestock (Bain et al, 2011). However, it is now acknowledged that this has significant detrimental impacts on vegetation, carbon stores and water resources (Tucker, 2003). The impact of burning on a peatland's hydrology may persist for a long time, and releases nutrients, making the habitat more eutrophic. Consequently, burning regulations in the UK no longer allow burning on peatlands.

Given that this is an EU scale study and that intentional burning has mainly occurred in the UK, and that many areas of burnt peat will recover naturally with time, we consider that this pressure has negligible impacts on the cost of achieving Target 2 and is not considered further here.

However, accidental and wildfires may still affect some mires, and especially those that are drier than normal due to drainage and peat extraction. The extent of fire risk is uncertain, but on going measures will be necessary to help prevent and control wildfires and accidental fires. Fires that are intense may cause a complete loss of vegetation which may result in long term erosion as discussed below.

Human-induced erosion and landslides
A certain amount of erosion, particularly of blanket peat, is part of the natural cycle of peatland (JNCC, 2011), but peat soils can be destroyed quickly. Following wildfires, erosion or severe overgrazing, upland peat can be left completely without vegetation. The surface of bare peat can rapidly dry out and become hydrophobic, and the dry peat particles are susceptible to erosion, which can destroy the peat soil very rapidly, exposing the underlying mineral substrate (Holden et al, 2004). In the UK, 16% of peatlands are severely eroded (Bain et al, 2011). In Ireland, human-induced erosion is threatening blanket bog (National Parks and Wildlife Service, 2008). Peatlands in the UK and Ireland are being affected by the construction of wind farms and communication and powerline infrastructures. Works associated with these infrastructures, including access roads or tracks, burial of cables or concrete foundations cause direct habitat loss but more importantly cause fragmentation and disruption of peatland hydrology (Stunell, 2010). In some sites this has led to peat landslides and erosion, and peat drying (National Parks and Wildlife Service, 2008). The Scottish Government has developed a good practice guide to avoiding landslides on peatlands[76].

The baseline rate and overall level of degradation from this pressure at an EU scale is unknown, but it is likely to be very low. For this study we assume that the overall level of degradation in 2010 from cumulative impacts of erosion are between **2% and 5%.**

Invasive species
Degraded bogs are frequently invaded by native nitrophilous species such as *Molinia caerulea*, *Betula pubescens*, *Eriophorum vaginatum,* and *Pteridium aquilinum*. They are also affected by invasive alien species that invade a range of wetland habitats (see section 6.10). It is not possible to quantify the extent of degradation due to invasive species in mires, however grazing and/or mowing management should control most invasive plant species.

Eutrophication and pollution
The eutrophication of the groundwater or surface water entering fens or bogs leads to changes in vegetation composition. The habitats are also affected by deposition of nitrates and sulphates from air pollution. Atmospheric pollution and changes in surrounding water inflows are particularly difficult to isolate as causal agents for degradation, yet they may have profound effects (Pakalne, 2004). In England it is estimated that 98% of blanket bog peatlands and 100% of raised bog peatlands are subject to levels of nitrogen deposition that are damaging to these habitats (Bain et al, 2011; Natural England, 2010).

Some management measures, such as grazing and mowing can help to reduce the impacts of eutrophication. However, at source measures to reduce emissions are most important and these are discussed in sections 5.2, 5.3 and 5.4.

[76] Scottish Government (2007) Peat Landslide Hazard and Risk Assessments: Best Practice Guide for Proposed Electricity Generation Developments http://www.scotland.gov.uk/Publications/2006/12/21162303/0

6.8.4 Key pressures and extent of degradation in 2020

Ecosystem loss to 2020

The protection of the remaining peatlands in Europe is expected to improve through the inclusion of emissions resulting from Land Use, Land Use Change and Forestry (LULUCF) in Member State's carbon budgets, together with an improved valuation of peatland carbon storage potential and an increasing carbon price. Other new policy or regulatory measures that can improve the protection of existing peatlands include public and commercial pressures are leading some to phase out the use of peat in the horticulture industry[77]. Incentives encouraging afforestation or conversion of peatlands to agriculture are expected to decline (assuming that markets for bioenergy avoid the production of biomass on carbon-rich soils through). There is also a proposed cross-compliance GAEC standard to prevent cultivation of carbon-rich soils as part of reforms of the CAP. This could have a significant impact on the conversion of low peatlands to agriculture. However, at the time of writing this report it seems unlikely that this GAEC standard will be included in the final 2014-2020 CAP regulation, and this measure is not therefore included in the reference scenario for this study.

Taking these factors into account it is anticipated that the rate of loss of mires will decline, but it is difficult to quantify the degree of change. Therefore a wide range of potential annual loss is assumed **from 0.05% to 0.1%** per year in 2020.

It is also important to note that the 40% to 60% of remaining peatlands that are already significantly degraded are likely to degrade to the point where they are completely lost as functioning peatlands unless restoration efforts are scaled up considerably within the next five years.

Key degradation pressures in 2020

It is assumed that further afforestation of mires will decline in part due to the low profitability of forestry on mires and stronger environmental protection measures. In particular, Sweden and Finland's future policies on afforestation of peat soils will play a critical role (Vasander et al, 2003). For example, in Sweden the only permitted form of forest drainage now is remedial ditching (ie ditch digging after clear-felling of a site in order to prevent the groundwater table from rising in basins and poorly drained flats); whereas in Finland, government funds still support maintenance ditching (ditch cleaning and improvement ditching) of previously drained areas in order to keep them drained (Paavilainen and Päivänen, 1995). It is, however, difficult to assess the actual rate of afforestation on mires in 2020 as baseline levels are also unknown.

This study also assumes that peat extraction will decline due to better protection, the need to reduce carbon emissions (with respect to peat combustion for energy) and, with respect to use in horticulture, increasing public awareness of its environmental damage and the

[77] Eg England target to phase out peat use by 2030 http://www.defra.gov.uk/food-farm/crops/peat/

development of compost with peat substitutes. However, insufficient information is available to quantify the likely rates of peat loss in 2020.

It is assumed that human-induced erosion and landslides will decrease due to better habitat protection under cross-compliance and other regulations, but will also be exacerbated by climate change effects on rainfall and flood events, so that degradation due to this factor may remain close to 2010 levels. Due to the possible role of climate change in exacerbating fires, but also better prohibition of burning through agricultural regulations, it is assumed that degradation due to wildfires will remain at negligible levels.

It is almost certain that the main pressure on mire ecosystems will remain lack of grazing and mowing due to increasing agricultural abandonment. In fact it is possible that abandonment may increase due to the declining profitability of extensive livestock farming. This may be exacerbated by possible cuts in Pillar 2 funding as a result of CAP reforms. On this basis we assume that **40-60%** of the area of mires may be degraded in 2020 due to inadequate grazing and therefore require annual maintenance support measures (including management of invasive plant species).

Overall degradation - 2020

As described above it is likely that some important pressures will decline by 2020. However, many of these pressures have long lasting and cumulative impacts and proactive restoration measures are usually needed to reverse their impacts. Some restoration is underway through for example agri-environment measures and LIFE projects but the scale of funding is relatively low and therefore restoration by 2020 will have little impact on overall degradation levels. We therefore assume in this study that overall degradation levels may fall slightly or remain the same, and on this basis adopt overall degradation levels in 2020 of between **75% to 95%.**

Table 6-33 Estimated baseline and expected 2020 percentage degradation levels (a) and conversion rates (b), and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5. % restoration required = expected degradation level minus the maximum level to achieve target.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020		
	Min %/y	Max %/y	Min %/y	Max %/y	NB
Overall	1.00%	1.20%	1.00%	2.20%	
Disturbed hydrology and drainage	Low?	Low?	Very low?	Very low?	
Afforestation	Low?	Low?	Very low?	Very low?	
Peat extraction	Low?	Low?	Very low?	Very low?	
Lack of grazing and mowing	0.800%	1.000%	0.800%	2.000%	
Overgrazing	0.000%	0.000%	0.000%	0.000%	1
Burning and wildfires	Low?	Low?	Low?	Low?	2
Human induced erosion	Low?	Low?	Very low?	Very low?	

Note: 1, In this study it is assumed that anticipated GAEC measures will reduce these pressures to acceptable levels. 2, Cost concerning intentional burning impacts are probably negligible at a EU scale,. But wildfires may be significant and increasing.

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels*1		Maximum level for 15% restoration		Expected in 2020		% restoration required in 2020*2		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	80.0%	95.0%	68.0%	80.8%	80.0%	95.0%	12.0%	14.3%	3
Disturbed hydrology and drainage	40.0%	50.00%	34.0%	42.5%	35.0%	50.0%	1.00%	7.50%	
Afforestation	15.0%	25.0%	12.7%	21.2%	15.0%	25.0%	2.25%	3.75%	
Peat extraction	2.00%	10.0%	1.70%	8.50%	2.00%	10.0%	0.300%	1.50%	
Lack of grazing and mowing	40.00%	50.0%	34.0%	42.5%	40.0%	60.00%	6.00%	17.5%	
Overgrazing	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.00%	0.000%	4
Burning and wildfires	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	2
Human induced erosion	2.000%	5.000%	1.70%	4.25%	1.000%	5.00%	0.00%	0.750%	

Notes: *1. Baseline degradation level minus 15%. *2. % restoration required = expected degradation level minus the maximum level to achieve target. **3.** Article 17 reporting and IMCG data. 2020 estimate includes impacts of climate change and eutrophication. 4. In this study it is assumed that anticipated GAEC measures will reduce these pressures to acceptable levels.

6.8.5 Key management and restoration measures for mires

Maintenance measures

When in good ecological and physical condition most types of mires require little management. In fact mires are sensitive to grazing and trampling, and therefore grazing is generally not recommended on intact mires (Scottish Natural Heritage, 2011; Šefferová et al, 2008). Fencing may therefore be necessary to exclude grazing animals. However, on partially drained sites, **extensive grazing** using traditional hardy breeds is necessary to prevent scrub encroachment. It is important not to overgraze rich fens or other mire sites as it can lead to serious degradation and soil erosion. A preliminary mowing is sometimes necessary before livestock can be let onto the area. In other cases mowing may be combined with grazing in order to eliminate invasive plants that herbivores refused to graze.

Mowing with light, small machinery adapted to the sensitive fen environment (low pressure tyres and twinned wheels) has been shown to be a successful management tool for restoring and maintaining species richness on fens (Scottish Natural Heritage, 2011; Šefferová et al, 2008). The use of heavy equipment destroys the mire vegetation and soil structure. Hand mowing with scythes and brush cutters may be necessary on small or sensitive sites or to control invasive species. Mowing late in the season is the primary technique for preserving species diversity and species composition on fens, but on abandoned areas currently dominated by *Molinia* grass or *Phragmites* reed, mowing earlier in the season is highly effective and can lead to rapid recovery of habitats irrespective of the length of time for which they've been abandoned (Šefferová et al, 2008). Mowing is also needed to control invasive species. Mowing every second year is usually sufficient in Sweden; mowing only half the area each year is recommended in Poland (Šefferová et al, 2008). Extremely wet fens with a predominance of moss communities, for example in Sweden, can be mown every 3-5 years (Šefferová et al, 2008). Biomass collection and removal from the site is necessary but very labour intensive. Late summer is best when habitats are less wet.

On some mires the management of water levels is feasible and beneficial. This may require regular on-going monitoring and the adjustment water inflows and outflows as necessary (eg through sluices or other water control structures).

The potential for restoration and re-creation

Mires must be capable of active peat growth to maintain their key ecological processes and biodiversity values. The key restoration measures must therefore restore the hydrological integrity of the peatland, restrict nutrient inputs, and restore peat-forming vegetation. There are boundary conditions beyond which mires cannot be restored, for example if too much of the peat has been destroyed, or if the hydrological changes in the peat are irreversible (Holden et al, 2004). Consequently it is important to note that much of the area that has been lost may not be or re-creatable. This is because the degree of degradation is the critical factor in determining whether an area can still be regarded as degraded mire or whether the habitat has been lost. Peatlands can be classified into five categories of state of degradation (JNCC, 2011):

- **active peatlands** characterised as likely to be peat-forming because of the quality of the vegetation cover and largely unmodified hydrology. This may include formerly degraded surfaces in cases where management has successfully restored near-surface water levels;
- **degraded peat** includes all intermediate stages between active and bare peat, where the peat retains a semi-natural vegetation cover, but with dominance either by graminoids (moor grassland) or ericoids (heathland) instead of peat-forming vegetation: restoration is usually feasible through rewetting and can be relatively rapid;
- **bare peat** has had all its vegetation removed (eg by erosion, peat extraction or fire) but has not been affected by a significant change of land use: restoration potential depends on the extent of hydrological modification, and the depth of the remaining peat in relation to the water table; restoration to active peat may no longer be possible;
- **archaic peat** may still have significant soil peat depth but is under other land use (eg cultivation): restoration to active peat is very difficult to impossible but the carbon store can be maintained through sympathetic land management;
- **wasted peat** has lost both its peat-forming vegetation and a significant depth of peat soil. It is generally not possible to restore. However, the remaining peat still represents an important carbon store, and these soils can be good places to restore other valuable vegetation types, such as semi-natural grassland (see section 0).

Therefore in this study we assume that mires cannot be re-created (or would not be re-created to achieve Target 2 due to its low cost-effectiveness) on wasted peat areas or archaic peat. We also assume that to achieve Target 2, mires need only be restored from afforested peatlands where drainage and the density and growth of trees has not been enough to result in significant loss of peat or the total loss of semi-natural mire vegetation.

Restoration measures

The main reasons for the degradation of bog and fen hydrology usually lie outside the core habitat area, and so **integrated catchment management** is needed. Blanket bogs especially depend to a large degree on maintenance of surface water flow patterns at a landscape scale and hence are particularly dependent on sensitive land management practices (Holden et al, 2004). However, small isolated mires and fens may require the creation of buffer zones around them, and in some situations the boundary may need to be sealed to prevent the loss of water. The restoration of such small isolated sites is often difficult and likely to be very expensive, and therefore unlikely to be commonly carried out to achieve Target 2.

The primary restoration goal is the re-establishment of a high water table (Holden et al, 2004; Šefferová et al, 2008). Water levels should be increased to soil surface level by **ditch blocking or filling**, usually by implementing a series of dams. The type of dams depends on the age of drainage system, depth of ditches, number of dammed places, access possibilities, and level of filling-in of drains. Peat dams (refills with strongly humified peat) or peat plugs with a polythene membrane or tin sheets are most commonly used. Peat and plastic ditch plugs are unsuitable for ditch blocking where slopes are steep and ditch waters scour around the plugs. Bales of local straw or heather allow water to flow through the bails, but slow the velocity and allow sediment to slowly accumulate.

Recovery of the water table in peatlands following ditch blocking can be relatively rapid. If enough of the key drainage channels are blocked, the natural regeneration of vegetation may fill remaining ditches with sediment without the need for active refilling. However, vegetation or hydrochemical recovery may not follow. Where drainage has resulted in changes to peat properties it is necessary to reconstruct the water storage capacity of the peat in order to allow *Sphagnum,* brown mosses, or other peat-accumulating species to regrow and accumulate dead plant material. Additional measures to increase water storage capacity include creating open reservoirs and/or using straw mulch.

Natural re-colonisation of vegetation is likely to be successful if the site's hydrology and nutrient status remains intact and if common habitat specialists are still present in the nearby vegetation and/or seed bank (Triisberg et al, 2011). The Rhynchosporion community develops naturally on cut-away areas of bogs with typically fluctuating water conditions (Stallegger, 2008), and in general the seeds and spores of bog species can survive for long periods in peat. If natural regeneration is impossible, inadequate or too slow, then proactive vegetation establishment may be needed. Most simply this may involve spreading *Sphagnum* diaspores or fragments across the surface of the bog, but there may be a need for additional protection by mulching. Mire species can be reintroduced both by plant transplantation and plant propagation from seeds. Rafting is the establishment of an initial layer of vegetation onto which a successional colonisation of other species take place, for example on bare peat. Bunds are constructed and the surface is flooded to encourage the re-colonisation of aquatic *Sphagnum* species, such as *Sphagnum cuspidatum*, to form floating mats. Eventually, it is hoped that successional colonisation of terrestrial *Sphagnum* species will take place.

Scrub clearance and removal of trees and rootstocks on overgrown mires generally needs to be done manually using a brushcutter or chain saw so as to avoid soil compaction and destruction of vegetation with machinery. Cutting and removal with heavy machinery should only be done with specialist low ground-pressure vehicles or where soil compaction is not a problem. Scrub and tree removal may even need to be done with the aid of a helicopter on sensitive raised bogs (eg LIFE project Restoration of Scottish Raised Bogs).

On heavily degraded peatlands the **removal of degraded peat** layers may be necessary to allow the underlying peat to absorb water and for vegetation to re-colonise, as well as removal of scrub and tree saplings (Triisberg et al, 2011). To restore fens it is often also necessary to remove the nutrient enriched top soil layer so that it can't mineralize in situ and induce a nutrient enrichment on the stand, either by removing the top soil completely or deep ploughing it and burying it under subsoil. Removal can be done by tarpaulins, corrugated iron or sleigh, which could be drawn out by horse or quad bike. These actions are costly, but can sometimes be partially reclaimed through revenue from selling the soil (Klimkowska et al, 2010a). Small-scale peat cuttings are very effective in providing and maintaining early succession stages, small pools, bare peat and low vegetation, and thereby diversifying vegetation composition and structure (Stallegger, 2008).

Lastly, mires within areas subject to human visitor pressure may require visitor management measures, including fencing, information provision, path and boardwalk construction. However, requirements for such measures cannot be reliably quantified and are therefore not costed in this study.

Table 6-34 below provides a summary of the key measures needed to maintain and restore the ecosystem and the percentage of the ecosystem affected by each pressure that it normally needs to be applied to.

Table 6-34 Key maintenance and restoration measures and their required percentage coverage to address key pressures

Degradation source /Key pressure	Integrated catchment management	Water-level management	Mowing, cutting	Extensive grazing	Fire prevention & control	Ditch blocking etc.	Scrub or tree clearance	Vegetation re-establishment of	Removal of degraded peat
Drainage and hydrological modification						100%		10%	10%
Afforestation	100%	5%				100%	100%	90%	50%
Peat extraction						50%		90%	
Burning and wildfires					100%			5%	
Human-induced erosion and landslides									
Lack of grazing / mowing, invasion of scrub & invasive species			5%	95%			5%		

6.8.6 Costs of management and restoration measures

Costs for restoration of mire habitats vary enormously between the different types of mire, from very high per hectare costs for core raised bog areas, to relatively low costs for degraded blanket bog or fen meadow. The main costs relate to the restoration and management of hydrology. Costs data was sourced for six countries including Ireland and the UK, but not for Sweden or Finland. This means we do not have costs data from two of the four countries with 93% of the EU's mire ecosystem.

From an analysis of all the cost data, the following conclusions can be drawn:

Combined costs for maintenance of mires range from 31 to 2,300 €/ha/yr. This includes payment for any or all of the following costs: measures to maintain a high water table throughout the year including blocking or maintenance of ditches; maintaining water control structures in good working order; cutting and removing tree or shrub growth; not digging or turning over peat; no fertilisers; monitoring, mapping and planning (GHK, 2006; Keenleyside et al, 2012; Natural England, 2009; Verheijen et al, 2009b). In the German federal state Mecklen-Vorpommern, a programme to fund mire maintenance is selling bonds to businesses at 5,000 €/ha (Schäfer, 2009), but it is not clear what measures this payment covers, so it is not used in these calculations. The lowest general maintenance costs for blanket bog in the UK (31 €/ha/yr) (GHK, 2006) and the highest for quaking bogs in the Netherlands (2,321 €/ha/yr) (Verheijen et al, 2009b) are taken as the band to work with in this study to assess general maintenance costs.

This study therefore assumes that an average overall maintenance management cost is **639 €/ha/yr**, based on an average of general maintenance management costs data from the Netherlands and the UK, and that this cost is incurred every year for a decade.

Data on specific maintenance costs include:

- Payments for annual **mowing** and removal of cuttings using light machinery, and no fertilisation, are 417 €/ha/yr in the Czech Republic (Keenleyside et al, 2012), but range from 5 to 1,802 €/ha/yr in the Netherlands (Verheijen et al, 2009b), with the level of payment depending on the type of bog or fen habitat. Here we use an approximate average of **400 €/ha/yr**.
- The only information on the cost of **grazing** mires could be found for raised bogs in the Netherlands where this is compensated for a rate of 18 €/ha/yr (Verheijen et al, 2009b). However, this is very low compared to the costs of grazing other semi-natural habitats and is unlikely to be typical. For example, the average cost estimate used in this study for extensive grazing of heathland and tundra ecosystems is 116 €/ha/yr. Taking this into account we assume a cost rate for mires of **100 €/ha/yr.**
- Ongoing payments for **removing trees and scrub** from bogs or fens average 677 €/infested ha (ranging from 18 to 1,312 €/ha in the Netherlands and the UK). However, controlling invasive scrub species (*Rhododendron ponticum*) is much more expensive, at 3,000 to 7,625 €/infested ha in the UK (The Scottish Government, 2012). An average of all costs for clearing scrub is 2083 €/ha. This study assumes that this will need to be carried out between two and four times in the next decade (Verheijen et al, 2009b), giving an annualised cost of **625 €/ha/yr**.
- Ongoing costs for managing the water table of bogs or fens are around **8 €/ha/yr** (sluice gate labour and maintenance costs).

*Combined restoration payments range from €512 to €1,083 in the UK, with an average of **735 €/ha**.* These payments cover costs for blocking ditches, digging out dried peat / fen cutting, implementation and management of a water management regime, scrub clearance, fencing, grazing (where appropriate) (GHK, 2006; Natural England, 2009; RSPB, 2011).

Capital costs for restoring mire habitats are much higher, some examples include:

- A project that restored 18,394 ha of blanket bog in Wales[78] cost £2.9 million, of which 55% was spent on practical work, therefore the cost of practical restoration work averaged 867 £/ha restored (1083 €/ha). Drain blockage cost an average of 2.8 £/m of blocked drains (using various different techniques), and tree removal had an upper cost of 1000 £/ha for chainsaw clearance at a tree density of 0.5 trees/m^2 (RSPB, 2011).
- An estimate of average costs for restoration of lowland raised bog, including capital and land costs, based on completed projects, was 4975 £/ha restored (6219 €/ha) (GHK, 2006).

[78] LIFE06 NAT/UK/000134

- A similar estimate for restoration of degraded blanket bog came to 500 £/ha restored (625 €/ha) (GHK, 2006).
- A study reports that the UK has spent an estimated £500 million over the last decade blocking drains on over 3200 ha of peatlands, which works out at an average cost of 15,625 £/ha/yr (19,500 €/ha/yr) (Ramchunder et al, 2012).

Specific restoration costs include:

- **Removal of degraded dried peat** from the surface of degraded bogs or fens costs in the order of **315 €/ha**, from as low as 4 to 541 €/ha depending on what kinds of techniques are used (eg sod cutting or choppering), again in the UK and the Netherlands.
- **Ditch or grip blocking** costs are between 69 and 250 €/ha restored (Environment Agency, 2010; Verheijen et al, 2009b), averaging approximately **170 €/ha**. Most projects only give this information per metre of ditch blocked, which is not possible to extrapolate to overall area restored. For example, a LIFE project in Ireland spent £30 per drain on blanket bog (LIFE02 NAT/IRL/8490) (Murphy, 2008). A LIFE project in the UK spent £1 to £5.5 per km of drain on blanket bog (LIFE06 NAT/UK/000134) (RSPB, 2011). The Scottish RDP compensates ditch blockage with plastic piling dams on wetland at £60 to £280 per dam (The Scottish Government, 2012).
- No specific information could be found on the costs of the **re-establishment of peat vegetation** in mires, and therefore for the purposes of this study we assume that they are equal to the cost of vegetation re-establishment on heathland and tundra ecosystems, which amounts to **75 €/ha/yr.**
- Similarly specific cost data for **tree and scrub clearance** on mires does not appear to be available. In heathland and tundra ecosystems these measures costs 100 €/ha/yr, but costs may be higher on mires due to the sensitive nature of the habitat and need to use specialist machinery or manual labour. Therefore we assume a cost of **150 €/ha/yr** for tree and scrub removal.

6.8.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration target. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). The costs of re-creating the ecosystem are not calculated because, as discussed above, once peat degradation has passed a certain point it is normally impossible or prohibitively expensive to restore a functional and sustainable mire ecosystem.

Two estimates of maintenance and restoration coats are provided based on two methods: firstly the overall level of ecosystem degradation and average combined costs of maintenance and restoration (Table 6-35), and secondly, the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures Table 6-36. See Chapter 3 for a detailed account of the methodology and its assumptions.

The estimated average additional annual cost of **maintaining** the ecological condition of mires is between €49 to 108 million, according to overall degradation rates and combined measures costs, or only between €8 to 10 million based on key pressures and the specific costs of key measures. The estimated average additional annual costs between 2010 and 2020 of achieving the **restoration target** is between €68 to 80 million, according to overall degradation rates and combined measures costs, or between €9 to 28 million based on key pressures and the specific costs of key measures.

It clear from the above results that the costs based on key pressures and the costs of key measures are much lower than those based on general degradation levels and combined measures. As discussed in Chapter 3, it is believed that the former method is normally likely to provide more reliable estimates. However, for this ecosystem a particularly large number of assumptions had to be made on pressures and the costs of specific key maintenance and restoration measures due to a lack of EU-27 wide data. Furthermore, mire ecosystems are degraded by a combination of the effects of damaged hydrology with the increasing impacts of eutrophication and climate change, and it is not realistic to separate out these impacts. The degradation impacts of past erosion, wildfires, and scrub invasion are closely linked to these key drivers. Therefore higher restoration costs calculated using the overall degradation levels may therefore be more realistic in this case.

It should also be noted that the costs of maintaining and restoring 15% of the degraded area are significantly influenced by what estimate is used for the total mire area in the EU-27. In this report, the CLC2006 area was used in order to provide a consistent dataset and to avoid double counting of areas, but it is likely that this significantly underestimates the area to be restored and the associated costs.

Table 6-35 Projected additional costs (€) in 2020 of achieving Target 2 in mires on the basis of the current costs of combined measures and expected overall degradation extent in 2020

Important notes: All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). See section 3.5 for further explanation. Ecosystem area = 7,675,305 ha

	Cost	Area required over		Cost in 2020*	
Activity	**€/ha**	**Min**	**Max**	**Min**	**Max**
Maintenance required in 2020*2	639	1.00%	2.20%	49,000,000	108,000,000
Restoration required 2010-2020	735	12.00%	14.25%	67,700,000	80,400,000

Table 6-36 Projected additional costs (€) in 2020 of achieving Target 2 within mires on the basis of current costs of key maintenance and restoration measures and expected key levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). See section 3.5 for further explanation. Ecosystem area = 7,675,305 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

	% Area at risk in 2020						Annual costs in 2020	
Key Pressure	**Min %/y**	**Max %/y**	**Key measure**	**Current annual costs (€/ha/y)**	**% area applied to**	**Instrument**	**Min**	**Max**
Disturbed hydrology and drainage	35.0%	50.0%	Integrated catchment management	0	100%	management planning		
			Water level management	8	5%	AEM	1,070,000	1,540,000
Lack of grazing and mowing	0.800%	1.000%	Mowing	400	5%	AEM	1,230,000	1,530,000
			Extensive grazing	100	95%	AEM	5,830,000	7,290,000
Over-grazing	0.000%	0.000%				GAEC	-	-
Burning and wildfires	?	?	Fire prevention & control	0	100%	regulation		
Human induced erosion	?	?			100%	regulation		
MAINTENANCE TOTAL							**8,140,000**	**10,400,000**

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% requiring restoration in 2020		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Disturbed hydrology and drainage	1.000%	7.500%	Ditch blocking	170	100%	AEM	1,300,000	9,790,000
			Vegetation re-establishment	75	10%	AEM	57,600	432,000
			Removal of degraded peat	315	10%	AEM	242,000	1,810,000
Afforestation	2.250%	3.750%	Ditch blocking	170	100%	AEM	2,940,000	4,890,000
			Scrub and tree clearance	150	100%	AEM	2,590,000	4,320,000
			Vegetation re-establishment	75	90%	AEM	1,170,000	1,940,000
Peat extraction	0.300%	1.500%	Removal of degraded peat	315	50%	AEM	363,000	1,810,000
			Ditch blocking	170	50%	AEM	196,000	979,000
			Vegetation re-establishment	75	90%	AEM	155,000	777,000
Lack of grazing and mowing	6.000%	17.500%	Scrub and tree clearance	150	5%	AEM	345,000	1,010,000
Human induced erosion	0.000%	0.750%	Vegetation re-establishment	75	50%	AEM	-	216,000
RESTORATION TOTAL							***9,360,000***	***28,000,000***

6.9 INLAND MARSHES

6.9.1 Description of inland marshes

Europe's freshwater ecosystems are rich in biodiversity and provide essential ecosystem services to the population, but are also heavily influenced and modified by human activities. Inland marshes are found on the floodplains along rivers and streams, along the margins of lakes and ponds, and in other low-lying areas where groundwater comes to the soil surface or where rainfall saturates the soil because drainage is slow. Inland marshes are low-lying, non-forested land which is usually flooded in winter and more or less saturated by water all year round (but do not include deep peatlands –see mires section 6.8). The inland wetland area is closely connected with lakes and rivers, as it includes all waterlogged riparian vegetation such as reed beds, rushes or sedges except swamp forest (see forests section 6.5) and humid grasslands that are only periodically waterlogged or only waterlogged below the soil surface (see semi-natural grasslands section 6.4). As these habitats are so closely associated, the restoration of inland wetlands is usually undertaken together their associated lakes, rivers, or salt lakes. However, in this study the inland marsh land cover (CLC 411) is discussed separately from rivers (CLC 511), lakes (CLC 512) and mires (CLC 412) in order to be consistent with the draft ecosystem typology being developed by the MAES working group (Maes et al, 2012).

6.9.2 Extent of inland marshes

A summary of the approximate area of inland marsh in each Member State is provided in Table 6-37. The CLC data show over 1 million ha of inland marsh (in the context of almost 12 million ha of freshwater habitat in the EU-27 in 2006). Half of this marsh area lies in Sweden and Finland. Other large areas of inland wetland lie in Romania (principally the Danube Delta), Poland (principally the Biebrza marshes), France, and Germany (see annex 1 for more details). Spain and Italy also historically had substantial areas of inland wetland, but have lost over 60%. There is however considerable uncertainty associated with mapping wetlands. Even in the well-documented UK, the area of wetland habitat is difficult to specify (Maltby et al, 2011).

This study uses the CLC area of inland marshes of 1,059,432 ha.

The CLC inland marshes area can be considered to be broadly related to a few of the habitat types listed in Annex I of the Habitats Directive (see annex 1 for further details)[79]. However, the Annex I habitat area and the CLC inland marsh cover cannot be considered to be fully equivalent, because most of the inland marsh CLC area is not defined in the Annex I habitat types (see annex 1 for more details). It is also important to note that Member States are only reporting a fraction of their freshwater ecosystem as Annex I habitat (Bensettiti and Trouvilliez, 2009).

[79] 15 other Habitats Directive Annex I habitat types including wet grasslands, some fen types, and alluvial forests, are related to inland marshes but not included here (see annex 1 of this report for details)

Table 6-37 Area of inland marshes CLC (hectares), and equivalent Annex I habitat area

The column shaded in grey denotes the area data that were used in this study. CLC data relate to 2006 CLC data except where indicated as * where data are for 2000. See annex 1 for list of Annex I habitat areas that correspond to CLC type 411. The Annex I habitat area does not include any areas in Romania and Bulgaria.

	CLC 411 Inland marshes (ha)	Annex I habitat area (ha)
Austria	20,816	1,500
Belgium	2,810	9,900
Bulgaria	8,973	n/a
Cyprus	524	0
Czech Republic	5,985	17,245
Denmark	29,260	370
Estonia	77,495	2,000
Finland	25,747	4,500
France	78,678	54,001
Germany	52,864	40,464
*Greece**	*140*	*138*
Hungary	76,322	201,500
Ireland	16,374	10
Italy	15,975	34,900
Latvia	23,382	1,600
Lithuania	19,208	4,400
Luxemburg		0
Malta		0
Netherlands	34,841	1,000
Poland	98,960	4,260
Portugal	1,135	0
Romania	336,300	n/a
Slovakia	2,692	2,645
Slovenia	2,438	6,600
Spain	54,315	42,120
Sweden	59,783	8,250
United Kingdom	14,415	351
TOTAL	**1,059,432**	**437,753**

6.9.3 Key pressures and extent of degradation – 2010 baseline

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures at the baseline year, which for this study is taken to be 2010. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Ecosystem loss

Inland wetlands are one of the most threatened ecosystems in Europe. Europe already lost over two-thirds of its wetlands before the 1990s, and between 1990 and 2006 a further 5% of marsh and bog area was lost (EEA, 2010a). Spain has lost more than 60% of its wetlands since 1970; France has lost 67% since 1950; Lithuania has lost 70% in the last 30 years (European Commission, 2007). The Rhine has lost over 90% of its floodplains in Germany, the Netherlands and Switzerland. Up to 80% of the total wetland area along the Danube and its larger tributaries, the Prut, Tisza, Sava, Drava, and Morava, have been lost (ICPDR, 2012). The rate of loss has slowed in the last decade; between 2000 and 2006 the CLC Land Cover change analysis[80] recorded a loss of 21,500 ha of inland marshes (CLC411) and mires (CLC412) together (whereas since 1990 126,700 ha have been lost) (EEA, 2010a).

Assuming the CLC land cover loss applies proportionally to inland marshes and mires according to relative area, it is assumed that at least 6800 ha of inland marsh was lost from 2000 to 2006, ie 0.9% of the 2000 area, or an annualised loss of **0.15%** over the period 2000 to 2010**.**

Key degradation pressures - baseline

Overall degradation - baseline
Taking into account recent assessments (ETC/BD 2008a; Tockner et al, 2009) and the specific pressures described below, this study assumes that the overall degradation rate of inland marshes lies between **50%** and **98%**.

Hydrological modification: drainage, separation of floodplain from river, low river flows, over-extraction of groundwater
The main cause of loss and degradation of inland marshes is drainage and hydrological modification, both for river and flood regulation, and for conversion of wetland to agriculture or forestry. Another cause of degradation due to hydrological modifications is the construction of dams for hydropower and resulting low river flow levels, or excessive water abstraction for agriculture or urban uses which starves the wetland of water. Mediterranean wetlands are particularly under pressure from over-abstraction of river and ground water for irrigation. For example, the Coto Doñana wetland in Spain has lost large areas of marsh grassland to scrub vegetation as a consequence of over-extraction of water (Muñoz-Reinoso, 2001 quoted in (EEA, 2009b). Large areas of wetland along the Mediterranean coastline are affected by saline intrusion into groundwater aquifers due to over-abstraction of groundwater for irrigation (EEA, 2009b). Over-abstraction of surface

[80] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

and/or groundwater combined with increasingly prolonged periods of drought and low rainfall due to climate change are causing reduced river flows, lowered lake and groundwater levels[81], and dried up wetlands (EEA, 2009c). It also reduces the ecosystem's capacity to absorb other pressures, such as pollution, fragmentation, sediment scarcity, and climate change, without serious ecological damage. The water exploitation index (WEI), which compares water abstraction with water availability at the national level (House of Lords, 2012). However, comprehensive data by catchments and river basin are not yet available, so these data do not necessarily reflect the extent and severity of over-exploitation of water resources in sub-national regions, or seasonal variation in water availability and water use (EEA, 2012c).

An intact inland marsh acts essentially as a store in the hydrological cycle, reflecting the balance between inputs (from river run-off, groundwater and rainfall) and outputs (evapotranspiration, river run-off, and transfer to groundwater). Marshes are dependent on processes operating at the scale of the whole upper catchment and/or aquifer recharge zone, and in turn influence the downstream habitats, estuary and coastal zone. Any degradation of a wetland's hydrological functions will have many knock-on effects on other ecosystems and habitats, as well as the adverse effects on the biodiversity and functioning of the wetland itself.

In this study it is assumed that **at least 10% and maximum 25%** of freshwater ecosystems are degraded due to low water levels and/or low groundwater levels.

Abandonment of extensive grazing and cutting of vegetation for traditional uses
Abandonment of traditional uses of marshes, including extensive grazing, reed cutting, and litter meadows, is causing scrub encroachment. Too much woody vegetation dries up small wetlands or chokes them with leaf litter. In this study it is assumed that **at least 30% and maximum 60%** of inland marshes are degraded due to lack of management.

Invasive species in freshwaters
European freshwaters have been invaded by a large number of alien species, some of which are causing large-scale disruptions. Generic costs of controlling invasive alien species are discussed in section 5.5.6.

Eutrophication from diffuse pollution
Although nitrate and phosphate concentrations in European freshwaters have improved significantly, some rivers still have excessive pollution levels, and eutrophication is a serious cause of damage in many wetlands, causing shifts in plant communities towards eutrophic vegetation (Ruiz, 2008). However, some wetland soils have a high capacity for denitrification, a capacity that is exploited in constructed wetlands. Measures to deal with the consequences of eutrophication are discussed in this section, at source control measures are discussed in sections 5.2, 5.3 and 5.4.

[81] Surface waters and groundwater are strongly interconnected, and rivers, lakes and wetlands may be strongly dependent on groundwater to provide critical base-flow levels when the surface flow is lowest.

6.9.4 Key pressures and degradation in 2020

Europe's freshwater ecosystems face significant pressures from economic development interests, including transport network plans[82], expansion of hydroelectric dams, increased use of artificial storage in reservoirs to generate peak demand electricity, demands for increased flood protection of urban areas, and increased water abstraction.

Climate change will also put increasing pressures on freshwater habitats, exacerbating both water scarcity and flooding (Wilby et al, 2006). Summers in the Mediterranean will become much drier, whilst heavy rainfall events during winter will become more frequent in northern Europe, increasing the incidence and severity of flash floods. Coastal freshwater wetlands such as reed beds and wet grasslands will be increasingly threatened by saline intrusion into groundwater and saline flooding from breaches of coastal dams and banks after storm damage, resulting from sea level rise (Natural England, 2008). Such freshwater habitats may be converted to saline habitats, resulting in the loss of freshwater species and feeding birds (but also providing an opportunity for the re-creation of salt marsh and meadow).

On the other hand, restoration of freshwater habitats provides significant opportunities for adaptation to climate change and for reversing some current negative impacts on biodiversity and ecosystem services. The potential for restoration and for creation of new wetland habitats is being increasingly realised. For example, Denmark's national wetland restoration programme has resulted in 2985 ha of new shallow lakes and 3844 ha with a variety of different wetland projects ranging from areas irrigated with drainage water to restored river valleys[83] (Hoffmann and Baattrup-Pedersen, 2007). These habitats mostly replaced arable or meadow grassland. Along the lower Danube, as of 2008, 469 km^2 of floodplain has been or is undergoing restoration, with a total restoration area of 2,236 km^2 in planning (Hulea et al, 2009).

For this study, it is assumed that the overall degradation of inland marshes in 2020 will still lie **between 50% and 98%,** whilst degradation due to low flows, drainage and water abstraction will increase to **15-30%.** The area requiring management will still be **30-60%.**

[82] For example, the Trans-European Networks for Transport (TEN-T) Corridor VII along the Danube aims to remove navigation bottlenecks from over 500km of free-flowing Danube stretches, containing the most ecologically valuable river habitats that remain. The proposed Danube-Odra-Elbe Canal would affect 46,000 ha of 38 protected areas. (RSPB et al, 2008; Tockner et al, 2009)

[83] The mean nitrogen removal for all projects was estimated to be 259 kgN/ha/yr, while results from the monitoring programme have shown that wetlands remove between 39 and 372 kgN/ha/yr.

Table 6-38 Estimated baseline and expected 2020 percentage degradation levels (a) and conversion rates (b), and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5. % restoration required = expected degradation level minus the maximum level to achieve target.

a) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Degradation rate 2010		Expected in 2020	
	Min %/y	Max %/y	Min %/y	Max %/y
Overall	**1.300%**	**2.000%**	**1.500%**	**3.000%**
Hydrological modification	0.500%	1.000%	0.500%	1.000%
Abandonment of grazing and cutting	0.800%	1.000%	1.000%	2.000%
Invasive species	?	?	?	?
Pollution	?	?	?	?

b) Annual degradation rates resulting from each key pressure

Degradation source / key pressures	Baseline 2010 levels		Maximum level for 15% restoration[*1]		Expected in 2020		% restoration required in 2020[*2]	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Overall	**10.0%**	**40.0%**	**8.5%**	**34.0%**	**20.0%**	**50.0%**	**11.5%**	**16.0%**
Hydrological modification	10.0%	25.0%	8.50%	21.25%	15.0%	30.0%	6.50%	8.75%
Abandonment of grazing and cutting	30.0%	60.0%	25.5%	51.00%	30.0%	60.0%	4.50%	9.00%
Invasive species	?	?			?	?		
Pollution	?	?			?	?		

Notes: 1. Baseline degradation level minus 15%. 2. % restoration required = expected degradation level minus the maximum level to achieve target.

c) Conversion rates

Conver sion causes	Conversion per year 2000-2010		Conversion per year 2010-2020		Total converted over 2000-2020		% re-creation required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Habitat loss	0.150%	0.150%	0.150%	0.150%	3.000%	3.000%	0.450%	0.450%
Total	**0.150%**	**0.150%**	**0.150%**	**0.150%**	**3.000%**	**3.000%**	**0.450%**	**0.450%**

6.9.6 Key measures for management and restoration

The major initiative to achieve a Good Ecological Status of freshwater ecosystems in the EU is the establishment of River Basin Management Plans under the Water Framework Directive. The Water Framework Directive covers all measures that will restore water quality and quantity, but does not necessarily include measures to restore riparian or wetland habitats and species that are not considered to be direct indicators of water status (see chapter 5). Effective conservation of freshwater resources requires integrated catchment management involving statutory controls, (agri-) environmental incentives and source targeted measures to reduce pressures from pollution and water abstraction. In the section below, it is assumed that the necessary maintenance and restoration activities required to maintain water status or to restore to achieve Good Ecological Status or Good Ecological Potential of freshwaters under the Water Framework Directive (WFD) are covered by Member States' baseline costs (see section 5.1). The following section therefore concentrates on measures for the maintenance and active restoration of hydromorphology and habitats of inland marshes.

Key maintenance measures for marshes

The maintenance of inland marshes needs to be in the context of the maintenance of the whole freshwater ecosystem, including the associated open water bodies (see section 6.10). Measures to maintain water levels include **sluice gate control and monitoring** and **ditch maintenance**. Because inland wetlands can be affected by agricultural run-off, it is often necessary to **maintain buffer zones** around the wetland.

Most inland marshes need periodic interventions to slow succession processes. Some inland marshes need to be maintained by **extensive grazing management** with livestock adapted to wet conditions. Alter grazing regime to light animals in late spring when conditions are drier in order to cover of more competitive species. Grazing should not be used during flooding periods to avoid soil compaction and trampling damage. The regular **removal of invasive species** is a widespread maintenance measure in the freshwater ecosystem.

Reed beds can to be managed by **reed cutting** every 2-3 years, which should be done in a mosaic /patchwork fashion so as to always leave a diversity of reed stands. Periodic **tree and scrub removal** in reed beds may be necessary. Reed bed decline is often related to eutrophication and water-level regulation, and requires at-source measures to reduce eutrophication (see section 5.1), and **restoration of water-level dynamics** that stimulate reed growth (see lakes section 6.10).

Key restoration measures for marshes

The restoration of inland marshes requires the restoration of hydrological conditions, which must be done in the context of the restoration of the whole freshwater ecosystem, including the associated open water bodies, such as large-scale projects to re-connect rivers with their floodplains (see section 6.10). Most of the measures specified here are therefore to be interpreted in the context of the measure described in section 6.10. **Raising water tables** in inland marshes requires measures to **block drainage channels** and ditches or **re-open ditches to the river**, **install sluices to allow controlled flooding**, and where silted, it may be necessary to **remove excess sediments**, restoring original/appropriate water depths.

Restoration of wetland vegetation may be possible by natural colonisation; seed banks of wetland species can resist years of desiccation. If the seed bank has deteriorated, **sediments** with seeds and propagules should be brought in from similar wetlands nearby. Habitats overgrown by shrubs and perennial plants may need **large-scale scrub clearance** with specially adapted machinery (wide-track cutters or floating cutters). In some areas it may be necessary to **clear scrub manually** during dry periods.

Table 6-39 Key maintenance and restoration and measures for inland marshes and their required percentage coverage to address key pressures

	Regulate / restore water levels	Maintain buffer zones	Extensive grazing	Cutting	Scrub removal	Reprofiling, ditch clearance/ blocking	Re-create freshwater habitat
Habitat loss							100%
Degradation due to low flows, drainage and water abstraction	100%	20%			10%	100%	
Abandonment of traditional management			40%	60%	20%	10%	

6.9.7 Costs of management and restoration measures

Costs of inland marsh maintenance

Inland marsh habitats vary greatly in their maintenance requirements and costs. Combined agri-environment payments for maintaining wetland areas including reed beds and marshes range go from 60 to 450 €/ha/yr in Germany, Finland, Italy, and the UK (Keenleyside et al, 2012; Natural England, 2009). Management of reed and wet marsh heath is costed at 2013 €/ha/yr in the Netherlands, whereas management of oligotrophic to mesotrophic standing waters or fens is 63 €/ha/yr (Verheijen et al, 2009b). Payments are for grazing, mowing reed, maintenance of water levels, banks and ditches, etc. Agreed costs for the extra management requirements of wetlands suffering from eutrophication in the Netherlands average 900 €/ha/yr, to a maximum payment of 1077 €/ha/yr (Oltmer et al, 2009).

Further costs include:

- the **maintenance of buffer zones** bordering wetlands is funded under agri-environment at 391-418 €/ha/yr in the UK and 480 €/ha/yr in Germany (Keenleyside et al, 2012)
- **Grazing** of wetlands, including maintenance of water levels, banks and ditches and control of invasive species, is funded at 262 €/ha/yr in Italy (Keenleyside et al, 2012)
- the **control of invasive species** in wetlands is compensated at 75 €/ha treated in the UK (Natural England, 2009)
- maintenance level **scrub removal** costs – no data

For the purpose of this study, it is assumed that the costs of maintenance of inland marsh are 200 €/ha.

Costs of inland marsh restoration

Costs for **reed bed restoration** lie between 75 and 1021 €/ha for digging out the bed and re-profiling ditches in the UK (GHK, 2006; Natural England, 2009). Restoration management of marshes is costed at an average of around 700 €/ha in the Netherlands, with a higher end cost of 1250 €/ha (Oltmer et al, 2009).

For the purpose of this study, it is assumed that the costs of restoration of inland marsh are 800 €/ha.

Costs of inland marsh habitat creation
Habitat creation in the form of floodplain scrapes in wetland in the UK cost an average of 1250 €/ha, whilst the creation of a wetland habitat mosaic through re-profiling cost over 93,000 €/ha (The River Restoration Centre, 2002). The creation of new reed bed in the UK costs 3500 - 6300 €/ha (DEFRA, 2002); however agri-environment payments in England for creation of reed bed are only 475 - 900 €/ha (Natural England, 2009) and 310 €/ha in Scotland.

For the purpose of this study, it is assumed that the costs of re-creation of inland marsh are 3000 €/ha.

6.9.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration and re-creation targets. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). See chapter 2 for a detailed account of the methodology and its assumptions.

Because of the scarcity of costs information that can be unambiguously assigned to inland marshes rather than freshwater habitats more widely (see section on rivers and lakes), it was not possible to complete an analysis of costs based on key pressures and the costs of specific key measures.

The results presented in Table 6-40 indicate that the estimated additional annual cost of **maintaining** the ecological condition of inland marshes is between €3 and 6 million. The average additional annual cost to 2020 of achieving the **restoration target** is between €10 and 14 million, whilst the average additional annual cost of achieving the **re-creation target** is approximately €1 million.

These cost estimates should be interpreted with caution because, data on degradation levels and costs of maintenance and restoration of inland marshes, as distinct from river and lake restoration, is scarce. It is also likely that the land cover data underestimates wetland area, as many wetlands are small and associated with other habitats.

Table 6-40 Projected additional costs (€) in 2020 of achieving Target 2 in inland marshes on the basis of the current costs of combined measures and expected overall degradation extent in 2020

NB: All costs are gross costs in Euros and based on current values using cost estimates from as close to 2012 as possible. Costs are all additional to existing (baseline) measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Costs do NOT include supporting actions, wide-scale / at source measures to address pollution impacts, or species-specific costs. See Chapter 2 for methodology.

	Cost	Area required over		Cost in 2020	
Activity	**€/ha**	**Min**	**Max**	**Min**	**Max**
Maintenance required in 2020	200	1.50%	3.00%	3,180,000	6,360,000
Restoration required 2010-2020	800	11.50%	16.00%	9,750,000	13,600,000
Re-creation required 2010-2020	2,000	0.45%	0.45%	953,000	953,000

6.10 FRESHWATER ECOSYSTEM – RIVERS & LAKES

6.10.1 Description of freshwater ecosystem

Europe's freshwater ecosystems are rich in biodiversity and provide essential ecosystem services to the population, but are also heavily influenced and modified by human activities. The EU has a number of major transnational river basins, as well as many other large river systems. The freshwater ecosystem includes Europe's large freshwater lakes, archipelagos of lakes, artificial freshwaters such as reservoirs, and inland salt lakes where evaporation results in progressive accumulation of mineral salt. Coastal lagoons are included in the coastal ecosystem (see section 6.11).

The restoration of rivers and lakes is closely related to their associated inland wetlands, including riparian and floodplain wetlands (see section 6.9). However, in this study the inland marsh land cover (CLC 411) is discussed separately from rivers (CLC 511), lakes (CLC 512) and mires (412) in order to be consistent with the draft ecosystem typology being developed by the MAES working group (Maes et al, 2012).

6.10.2 Extent of rivers and lakes

The total area of river and lake ecosystems

A summary of the approximate areas of water courses and water bodies[84] in each Member State is provided in Table 6-41. The CLC data suggest that there are over 10.6 million ha of freshwater habitat in the EU-27 in 2006. According to these data, some 91% of the ecosystem area is in water bodies and 9% in water courses, of which two-thirds lies in Sweden and Finland (35% and 30% respectively).

It is important to note that the measurement of the area of rivers and lakes is difficult due to technical problems (such as remote sensing resolution limitations), seasonal variations in the presence and size of water bodies and varying definitions of what constitutes a river and lake etc. **Consequently, CLC river and lake data are probably not reliable and need to be interpreted carefully.**

Firstly, Europe has numerous rivers, streams, ditches and canals that are narrower than 100 m, and small lakes and ponds that are under 20 ha, but these are not included in the CLC freshwater area. The CLC freshwater area is therefore likely to be a significant underestimate of the area of freshwater ecosystem in most Member States. For example, Denmark's agricultural grassland landscape is criss-crossed by about 35,000 km of natural watercourses and 25,000 km of artificial ditches and channels (Normander et al, 2009). Similarly, the UK has about half a million ponds, a figure which includes some turnover – around 18,000 ponds were lost and around 70,600 ponds were gained between 1998 and 2007 (Natural England, 2011). As regards typical ditches and ponds, this is in fact not a problem because such features are included in the area of agricultural land and other ecosystems. Furthermore, their maintenance and restoration is typically dealt with through land use measures (such as CAP cross-compliance regulations or agri-environment

[84] Note that the use of "water body" by CLC is not equivalent to the use of the same term under the WFD.

schemes), so in fact this omission from the river and lake ecosystem type is actually appropriate.

Nevertheless, there are likely to be more serious underestimates of larger rivers and waterbodies, as illustrated by examination of Habitats Directive and WFD data sources on the extent of these ecosystems. The CLC river and lake ecosystem area can be considered to be broadly related to the river and lake habitat types listed in Annex I of the Habitats Directive (see annex 1 for further details), although the Annex I habitat area and the CLC freshwater are not equivalent because the Habitats Directive freshwater habitat types consider the ecosystem of water and riparian vegetation together (and this reported area is likely to register as a complex mix of different CLC classes, including some grassland and forest – see annex 1 for more details)[85]. Under the Habitats Directive, Member States generally only appear to be reporting a fraction of their freshwater ecosystem as Annex I habitat eg (Bensettiti & Trouvilliez, 2009). However, Austria, Denmark, Finland, Greece, and Slovenia have reported areas of Annex I freshwater habitat that are significantly larger than their CLC freshwater area. For example, Austria has reported 978,800 ha of habitat 3220 'Alpine rivers and the herbaceous vegetation along their banks', as well as 8,050 ha of other river habitat, but the CLC river cover for Austria only totals 21,971 ha. Slovenia has reported some 9,000 ha of habitat 3150 'Natural eutrophic lakes with Magnopotamion or Hydrocharition-type vegetation', and more than 4,000 ha of other lake habitats, but the CLC lake cover for Slovenia only totals 2,745 ha.

Member States are obliged to report on the status of their river and lake water bodies under the WFD (though they use different cut off points to exclude small water bodies from their reporting). Up to 2012, Member States have reported a total river length of 1.17 million km (EEA, 2012d). If it is assumed that the CLC river area of 1,004,227 ha applies to rivers of at least 100 m width, the CLC area is equivalent to a maximum length of 100,423 km. This reinforces the conclusion that the CLC river area is a substantial underestimate of Europe's rivers.

On the other hand, WFD and CLC data seem to be more consistent with respect to the area of lakes. Up to 2012, Member States have reported more than 19,000 lake water bodies covering an area of 8,800,000 ha (EEA, 2012d), which is a little less than the CLC area of 9,680,195. The two lake ecosystem areas cannot be directly compared due to the bases of classification of the two approaches.

Despite the limitations and uncertainties associated with the CLC data on rivers and lakes, these are used in this study to facilitate comparisons with other ecosystems in this study, and because other data sets also have their own limitations. Based on the CLC data this study assumes that the total area of rivers and lakes in the EU is 10,684,422 ha, and within this the area of rivers is 1,004,227 ha.

[85] 15 other Habitats Directive Annex I habitat types including wet grasslands, some fen types, and alluvial forests, are related to inland marshes but not included here (see annex 1 of this report for details)

Area of heavily modified water bodies and artificial water bodies

The WFD allows Member States to identify **heavily modified water bodies,** which have modifications in their physical structure to serve various uses (eg navigation, flood protection, hydropower and agriculture) that are not feasible or desirable to remove[86], and **artificial water bodies** such as canals, reservoirs or open-cast mining lakes. These water bodies are considered to be restorable to a status of Good Ecological Potential rather than a nearly natural undisturbed condition (Kampa and Laaser, 2009). This corresponds to a lower level of restoration than is envisaged to meet Target 2 for other rivers and water bodies, and other semi-natural habitats.

Around 10% of lakes and 13% of river water bodies have been identified as heavily modified, and more than 6% of lakes and around 4% of river water bodies as artificial, but the designation process is still not complete (particularly in Denmark and Italy), and 10% of rivers and 0.3% of lakes are not yet designated.

In this report we assume that around **20%** or 2,136,884 ha of the CLC lake and river area is heavily modified or artificial, and therefore subject to a different standard of restoration and maintenance than the rest of the freshwater ecosystem.

It is important to note that Member States differ markedly in the proportion of their water bodies declared as heavily modified, and therefore in the degree to which their restoration efforts under Target 2 will correspond to the lower or higher restoration targets (see Chapter 2 for details). The Netherlands, Belgium, Hungary and Germany have the highest proportion of river water bodies designated as heavily modified or artificial. Other Member States such as Finland, France, Slovakia, Sweden and Ireland designated only 5% or less of their river water bodies as heavily modified or artificial. In the case of lakes, the highest proportions (above 60%) designated as heavily modified or artificial are in Belgium, the Czech Republic, the Netherlands, Bulgaria, France, the UK, Hungary and Italy. The Member States designating less than 5% of their lakes as heavily modified or artificial are Sweden, Estonia, Latvia, Ireland and Finland (EEA, 2012d).

[86] Heavily modified water bodies are those in which the changes to the hydromorphological characteristics of that body which would be necessary for achieving Good Ecological Status would have significant adverse effects on the wider environment and/or the essential services provided by the modified water body to sustainable human development (WHERE those essential services provided by modified characteristics of the water body cannot, for reasons of technical feasibility or disproportionate costs, reasonably be achieved by other more environmentally beneficial means). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for the Community action in the field of water policy, Article 4(3)(a) and (b).

Table 6-41 Areas of rivers and lakes CLC (ha), and Annex I freshwater habitat area

The column shaded in grey denotes the area data that were used in this study. CLC data relate to 2006 CLC data except where indicated as * where data are for 2000. See annex 1 for list of Annex I habitat areas that correspond to CLC types 511 and 512. The Annex I habitat area does not include any areas in Romania and Bulgaria.

	CLC 511 Water courses (ha)	CLC 512 Water bodies (ha)	TOTAL water courses and bodies (ha)	Annex I habitat area (ha)
Austria	21,971	46,663	68,634	1,029,850
Belgium	5,817	10,560	16,377	1714
Bulgaria	30,489	64,939	95,428	n/a
Cyprus	27	1,562	1,589	20
Czech Republic	4,450	51,962	56,412	10,124
Denmark		38,167	38,167	45,250
Estonia	3,889	209,860	213,749	48,050
Finland	69,550	3,090,324	3,159,874	3,384,030
France	118,363	214,565	332,928	179,360
Germany	74,561	332,398	406,959	287,959
Greece*	630	564	1,194	9,666
Hungary	46,267	129,349	175,616	56,030
Ireland	7,718	117,572	125,290	264,755
Italy	47,894	173,285	221,179	93,800
Latvia	14,882	104,493	119,375	82,430
Lithuania	17,891	106,354	124,245	40,890
Luxemburg	355	684	1,039	0
Malta			-	78
Netherlands	46,136	269,130	315,266	11,100
Poland	76,613	386,877	463,490	445,281
Portugal	19,918	53,102	73,020	15,440
Romania	208,096	167,485	375,581	n/a
Slovakia	10,393	21,186	31,579	2,517
Slovenia	5,159	2,745	7,904	28,110
Spain	45,923	248,411	294,334	139,032
Sweden	122,660	3,622,367	3,745,027	1,222,000
United Kingdom	4,575	215,591	220,166	167,857
TOTAL	**1,004,227**	**9,680,195**	**10,684,422**	**7,573,064**

6.10.3 Key pressures and extent of degradation – 2010 baseline levels

Many of the EU's rivers and lakes are degraded as a result of a combination of pressures, including pollution from agriculture, sewage and industrial effluents; water abstractions, river engineering for navigation and flood prevention, and disturbance (EEA, 2010a). However, as described in sections 5.2 and 5.3, progress is being made with tackling some of these pressures, in part as a result of actions under the Urban Waste Water Treatment Directive, the Nitrates Directive and most recently the WFD. Indeed, as described in section 5.1 it is considered in this study that the WFD targets for river and lake restoration and maintenance are broadly equivalent to Target 2 goals. Under the Directive Member States are required to ensure river and lake water bodies have a **Good Ecological Status**, unless they are heavily modified or artificial water bodies[87], in which case they need only be restored to **Good Ecological Potential**[88] by 2015. It is important to note that Good Ecological Status requirements must take into account the ecological conditions needed to achieve Favourable Conservation Status of water bodies that equate to habitats of Community interest. There is also a requirement to avoid deterioration of water bodies, which can be viewed as being equivalent to the need to maintain ecosystems under Target 2.

As described in section 5.1, although recent reports on the implementation of the WFD indicate that progress has been slow it seems highly likely that measures to reduce pressures covered by the WFD and reverse degradation under the WFD by 2020 will be more than sufficient to achieve the restoration component of Target 2. However, although we can assume with some certainty that declines in pressures and overall degradation will be sufficient to achieve the 15% target for most ecosystems it is not possible with the available information to reliably quantify future 2020 levels. Therefore, the following two sections outline some of the existing pressures that are expected to be addressed by the WFD, but we do not attempt to provide rough estimates for these pressure levels, because they are not required to continue with the estimation of the overall costs of meeting Target 2 for rivers and lakes. The analysis of the baseline status is primarily based on Member States reporting on the ecological status of lake and river water bodies under the WFD up to 2012 (EEA, 2012d; ETC/ICM, 2012a; ETC/ICM, 2012b).

However, it is important to note that the WFD only requires Member States to restore and maintain riparian habitats in so far as this affects the Good Ecological Status or Good

[87] Heavily modified water bodies are those in which the changes to the hydromorphological characteristics of that body which would be necessary for achieving Good Ecological Status would have significant adverse effects on the wider environment and/or the essential services provided by the modified water body to sustainable human development (WHERE those essential services provided by modified characteristics of the water body cannot, for reasons of technical feasibility or disproportionate costs, reasonably be achieved by other more environmentally beneficial means). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for the Community action in the field of water policy, Article 4(3)(a) and (b).

[88] A water body shows Good Ecological Potential if its biological quality elements (same as for Good Ecological Status) differ only slightly from the reference condition for heavily modified water bodies, which is intended to describe the best approximation to a natural aquatic ecosystem that could be achieved given the hydromorphological characteristics that cannot be changed without significant adverse effects on the specified use or the wider environment (CIS 2003 quoted in (Borja and Elliott, 2007)).

Ecological Potential of the water body, but does not require their restoration or maintenance if this has no influence on water quality. Therefore we anticipate that some pressures will remain, principally being loss of bankside habitats and connectivity with floodplains, and non-aquatic invasive alien species. The WFD also does not cover vertebrate species other than fish, including measures for mammals dependent on freshwater habitat. The two sections are therefore followed by an analysis of these pressures on rivers and lakes because additional measures will need to be taken to address them to achieve Target 2.

Ecosystem loss

The CLC area of water courses and water bodies increased by 59,200 ha (1.7%) between 2000 and 2006[89] (EEA, 2010a). Therefore this study assumes no overall habitat creation from the baseline. However, it is important to note that the increase is mainly due to an increase in artificial water bodies such as reservoirs, and hides the loss of smaller water bodies such as ditches and ponds from Europe's agricultural landscape (Farmer et al, 2008). It is important to note that intact freshwater habitats are key ecosystem service providers and biodiversity hotspots in the landscape irrespective of their size.

Overall degradation of rivers and lakes - baseline

Under the WFD, Member States reported that 44% of lake water bodies and 64% of river length does not have a Good Ecological Status or Good Ecological Potential. 14% of lake and river water bodies have unknown ecological status[90]. Most available quantitative information on status and degradation of rivers and lakes is given either for a whole river sub-basin or water body, or per km of river length, and it is not possible to convert these data into hectares of freshwater land cover.

According to Member States reporting of Annex I freshwater habitats, 64% of assessments were in unfavourable conservation status in the EU-25 in 2007. However, around 15% of the assessments are unknown, including most of Spain's Atlantic and Mediterranean freshwater habitats, oligotrophic lakes in Portugal, many of Austria's rivers, and many of Slovakia's freshwater habitats, plus the assessment did not include the substantial freshwater habitat area in Romania and Bulgaria (EEA, 2010a). In addition, as stated in annex 1, there is not a clear overlap between the Habitats Directive Annex I habitat area and the CLC area for freshwater ecosystems. It is therefore not clear that the Article 17 assessments provide a reasonably representative and reliable indication of the overall level of degradation of the ecosystem type.

Specific key pressures on freshwater ecosystem covered by the WFD

[89] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

[90] mainly in Poland (79%), Finland (54%), Italy (48%), Hungary (39%) and Greece (38% unknown).

It is difficult to ascertain the percentage of river and lakes ecosystems that are currently degraded as a result of specific key pressures as EU-wide information is lacking and/or inconsistent.

Chemical and biological pollution in rivers and lakes
Although nitrate and phosphate concentrations in European freshwaters have improved significantly, some rivers and lakes still have excessive pollution levels (Ruiz, 2008). Nitrogen and phosphorus enrichment causing eutrophication is affecting around 30% of rivers but less than 20% of lakes. Acidification from air pollution is affecting nearly 10% of rivers and around 17% of lakes (though only 10 Member States reported this impact on rivers and seven on lakes). Organic enrichment due to pollution by oxygen-consuming substances is reported to affect around 15% of rivers and 10% of lakes.

Although water quality has improved during recent decades, contamination with pesticides, pharmaceuticals and other persistent chemicals, particularly endocrine disruptors, is still a major concern in aquatic ecosystems.

Measures to deal with the consequences of pollution are discussed in this section, at source control measures are discussed in sections 5.2 and 5.3.

Hydromorphological pressures affect almost half of the river water bodies and 25% of lake water bodies reported under the WFD (EEA, 2012d). They can be subdivided into impacts related to reduced water levels (water abstraction), modified water flow regimes, fragmentation of upstream-downstream connectivity, changed river channel dynamics, and loss of lateral connectivity and riparian habitats due to river and lake engineering. 'River management' pressures reported under the WFD include physical alteration of the river channel including the effects of dredging, land drainage and barriers due to bridges, culverts etc. (ETC/ICM, 2012a). Countries with a high proportion of river and lake water bodies being affected by hydromorphological pressures are found in Central Europe towards North Sea as well as the Baltic Sea (ETC/ICM, 2012a).

Reduced water levels in rivers and lakes
Over-abstraction of surface and/or groundwater combined with increasingly prolonged periods of drought and low rainfall due to climate change are causing reduced river flows, and lowered surface (lake) and groundwater levels[91] (EEA, 2009c). It also reduces the ecosystem's capacity to absorb other pressures, such as pollution, fragmentation, sediment scarcity, and climate change, without serious ecological damage. Water abstraction and drainage have led to smaller water bodies shrinking or drying up, which can be particularly harmful where water bodies form part of wetland ecosystems. For example, special habitats like turloughs have suffered from drainage (JNCC, 2012). The water exploitation index (WEI) compares water abstraction with water availability at a regional level. However, comprehensive data by catchments and river basin are not yet available, so these data do not necessarily reflect the extent and severity of over-exploitation of water resources in sub-national regions, or seasonal variation in water availability and water use (EEA, 2012c).

[91] Surface waters and groundwater are strongly interconnected, and rivers, lakes and wetlands may be strongly dependent on groundwater to provide critical base-flow levels when the surface flow is lowest.

Modified water flow regimes in rivers

Most of Europe's rivers have a heavily modified flow regime due to river channelization that leads to faster flows, interbasin diversions, and to water storage reservoirs and/or hydroelectric dams. This has a series of negative impacts on the river habitat, flora and fauna downstream, including minimum flows that are too low to sustain biodiversity, and irregular flashy water regimes (hydropeaking) that increase flooding. In addition, large storm overflows from cities along a river can lead to abrupt increases in water flow and sediment load. As a very rough estimate, only some 2% of the freshwater river ecosystem in the EU-27 is still hydrologically intact with meanders, braiding, oxbow lakes, gravel banks, sand bars, and other features (Tockner et al, 2009). Only a very few rivers in the EU-27 are considered to be free-flowing (and therefore relatively unaffected by hydrological modifications) – these are some Swedish rivers and some small rivers such as the Amendolea in Italy, or the Frome and Piddle in the UK[92] (Tockner et al, 2009). On major rivers, only very short sections still remain relatively free-flowing. Around 30% of the Danube's mainstem length is impounded, and more than 80% of its length is regulated for flood protection (ICPDR, 2012). At least 95% of the Rhine is regulated, and the Alpine and Upper Rhine is strongly affected by hydropeaking (Tockner et al, 2009). Most Spanish rivers show significant alterations in their natural Mediterranean pattern, having lower discharges in winter when water is being stored in reservoirs and higher discharges in summer when water is released for irrigation (González del Tánago et al, 2012).

Fragmentation of upstream-downstream connectivity for freshwater river species

Dams, weirs and hydropower facilities result in reduced or completely lost connectivity for migrating fish and other aquatic species (EEA, 2010a). These structures also often create barriers to the migration of beavers and otters, as well as smaller freshwater animals. For example, the Danube river system has 18 major dams of which only 2 are passable for fish, as well as the numerous smaller dams and weirs on tributaries; of the 30 native fish species of the Danube river system, two species have gone extinct and 25 are globally threatened (Tockner et al, 2009).

Reduced river channel dynamics and diversity: incised channels, loss of river bed habitats, altered sediment dynamics

The modification of river channels has generally led to faster flow regimes, which leads to less groundwater penetration, increased scouring and an incised channel. Instream boulders and wood debris, which form vital river habitats, are removed to facilitate navigation, or are washed away. The Rhine and the upper Danube have lost almost all of the vegetated islands within their river beds, although the lower Danube still has many. Dams retain sediment, resulting in scouring downstream due to the loss of sediment. For example, the lower part of the Ebro river in Spain transports only 1% of the sediment it transported a century ago, due to dams trapping sediment (Tena et al, 2011). In numerous places continual replacement of sediments is necessary to counter this erosion, for example on the Upper Rhine downstream of the Iffezheim power plant around 180,000 m^3 of gravel must be added annually to the river to prevent further channel degradation, whereas along the

[92] However, the Piddle is heavily affected by water abstraction which has reduced the brown trout population (EEA, 2009b)

Middle Rhine about 30,000 m^3 of sediment must be removed annually to keep the channel open (and similar actions are required at other places along the river) (Tockner et al, 2009). Especially problematic are when large amounts of stored sediment are flushed down the river from a dam at periodic intervals.

Invasive species in freshwaters
European freshwaters have been invaded by a large number of alien species, some of which are causing large-scale disruptions. About 30 fish species have been introduced into the Danube, more or less outnumbering the native species diversity (Tockner et al, 2009). Canals such as the Rhine-Main-Danube canal connect up Europe's freshwater network and provide migration routes across the continent. The Asian freshwater clam (*Curbicula fluminea)* and zebra mussel (*Dreissena polymorpha*) are found along almost the whole Danube, and in the Middle and Lower Danube they are locally the dominant macro invertebrates (Tockner et al, 2009). Coypu (*Myocastor coypus*) and the signal crayfish also cause a lot of damage. The WFD requires Member States to eradicate aquatic invasive species "if practicable".

Specific key pressures on freshwater ecosystem not addressed by the WFD - baseline

Invasive species in riparian habitats
The WFD does not cover terrestrial invasive species that invade riparian habitats if they do not significantly influence water quality, including the particularly damaging Himalayan balsam (*Impatiens glandulifera*) and Japanese knotweed (*Fallopia japonica*).

In this study it is assumed that **1 to 2%** of riparian habitats (rivers and lakes) are affected by terrestrial invasive species, and that the control of these species is not required under the WFD if it does not affect the ecological status of the water body itself. This also applies to the area of heavily modified ecosystem.

River engineering: loss of lateral connectivity and riparian habitats including floodplains
Most of Europe's rivers are heavily modified by river bed straightening, embankments, and canalisation. This engineering has disconnected rivers from their floodplains, secondary channels, and other riparian habitats (see below under key pressures on marshes). Loss of connectivity with floodplains results in loss of biodiversity and productivity, greater force of flood waves, decreased water quality because of lack of sediment and nutrient trapping, and lower groundwater recharge rates. In the UK and Ireland, over 15% of river lengths[93] are separated from their floodplain by embankments (Maltby et al, 2011). Under the WFD, over 40% of river water bodies were reported as affected by altered habitats (ETC/ICM, 2012a). About half of the Member States reporting under WFD had more than 40% of their river water bodies affected by altered habitats and Hungary and Germany had more than 60% of their river water bodies being subject to impact from altered habitats. At the landscape scale, very few braided river systems are left in Europe (Tockner et al, 2009).

Overall it seems reasonable to assume that between 30% and 60% of the river area is degraded due to river engineering that has disconnected rivers from their riparian habitats,

[93] Regular sampling sites at every 100 m along the water course

including floodplains. However, as discussed in section 3.2.4, this study does not expect restoration of the fundamental abiotic conditions of heavily modified or artificial water bodies. Therefore this pressure and form of resulting degradation does not apply to these water bodies, which as noted above, make up some 17% of rivers. As such water bodies typically have altered structures, then we assume for this study that (with some rounding) this pressure only requires restoration on between **10% and 45%** of rivers.

Lake engineering: loss of lateral connectivity and riparian habitats including floodplains
Under the WFD, 20% of lake water bodies are reported as affected by altered habitats, and 30% by hydromorphology. However, it should be noted that the extent of degradation varies greatly and assessments may also not be consistent. For example, in the Czech Republic, UK and Hungary more than 60% of lake water bodies have been reported as having significantly altered habitats (ETC/ICM, 2012a).

Taking the WFD reports as the most consistent data source, we assume that at least 20% and at most 30% of the lake area is degraded due to loss of lateral connectivity and riparian habitats including floodplains. However, as discussed for rivers above, this pressure does not apply to these water bodies. Therefore as 16% of lakes are heavily modified or artificial we assume for this study that (with some rounding) this pressure only requires restoration on between **5% and 15%** of lakes.

Recreation and tourism pressure
Recreational activities in water bodies can pose a threat by damage to river shores through erosion and construction of structures, and to lake shores through increased wave erosion, and causes disturbance to resting or foraging waterfowl. More widely tourism pressures results in increased localised urban development. However, it is not possible to quantify the degradation from recreational and tourism activities. Therefore (as with other ecosystems) the costs of alleviating human disturbance related impacts are not estimated in this study.

6.10.4 Key pressures and degradation in 2020

As described above, many rivers and lakes are degraded in the EU, and consequently major actions and measures are already needed to protect and sustainably manage them (Tockner et al, 2009). Furthermore, Europe's freshwater ecosystems are likely to face increasing pressures from economic development interests, including transport network plans[94], expansion of hydroelectric dams, increased use of artificial storage in reservoirs to generate peak demand electricity, demands for increased flood protection of urban areas, and increased water abstraction. Climate change will also put increasing pressures on freshwater habitats, exacerbating both water scarcity and flooding (Wilby et al, 2006). Climate change is projected to decrease annual river flow in southern and south-east Europe, increase it in northern and north-east Europe, strongly alter seasonality of river flows, and increase the intensity and frequency of floods (EEA, 2009b). Summers in the Mediterranean will become much drier, whilst heavy rainfall events during winter will become more frequent in northern Europe, increasing the incidence and severity of flash floods.

[94] For example, the Trans-European Networks for Transport (TEN-T) Corridor VII along the Danube aims to remove navigation bottlenecks from over 500km of free-flowing Danube stretches, containing the most ecologically valuable river habitats that remain. The proposed Danube-Odra-Elbe Canal would affect 46,000 ha of 38 protected areas. (RSPB et al, 2008; Tockner et al, 2009)

On the other hand, the seriousness of the situation is recognised in the WFD and, as described in Section 5.1, progress is being made with tackling the most of the major pressures. Furthermore, the potential for restoration and for creation of new wetland habitats is being increasingly realised. For example, Denmark's national wetland restoration programme has resulted in 2,985 ha of new shallow lakes and 3,844 ha with a variety of different wetland projects ranging from areas irrigated with drainage water to restored river valleys[95] (Hoffmann & Baattrup-Pedersen, 2007). These habitats mostly replaced arable or meadow grassland. Along the lower Danube, as of 2008, 469 km^2 of floodplain has been or is undergoing restoration, with a total restoration area of 2,236 km^2 in planning (Hulea et al, 2009). Such restoration provides significant opportunities for adaptation to climate change and for reversing some current negative impacts on biodiversity and ecosystem services.

It is, however, very difficult to predict what degradation levels may be in 2020 and therefore, as described above, we do not attempt to estimate future degradation levels for pressures that are expected to fall by at least 15%; and therefore meet Target 2 requirements. Table 6-42 below therefore only provides estimates of those pressures that will require additional measures to be taken to achieve Target 2, namely the loss of riparian habitats and floodplain connectivity, and terrestrial invasive species. As adequate information on the likely future drivers of these pressures and possible policy impacts (eg from indirect effects of the WFD) is not available, **we assume in this study that the pressures will remain unchanged to 2020**.

[95] The mean nitrogen removal for all projects was estimated to be 259 kgN/ha/yr, while results from the monitoring programme have shown that wetlands remove between 39 and 372 kgN/ha/yr.

Table 6-42 Estimated baseline and expected 2020 percentage degradation levels and corresponding requirements to achieve a 15% restoration target

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5. % restoration required = expected degradation level minus the maximum level to achieve target.

Degradation source / key pressures	Baseline 2010 levels		Maximum level to achieve 15% restoration target		Expected in 2020		% restoration required in 2020		NB
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %	
Overall	**15.0%**	**60.0%**	**12.7%**	**51.0%**	**15.0%**	**60.0%**	**2.25%**	**9.00%**	**1**
Reduced water levels		?		?		>15% below baseline		None	2
Pollution		?		?		>15% below baseline		None	2
Modified water flows		?		?		>15% below baseline		None	2
Fragmentation of river connectivity		?		?		>15% below baseline		None	2
Reduced river channel dynamics and diversity		?		?		>15% below baseline		None	2
Loss of riparian habitats and floodplain connectivity (rivers)	10.0%	45.0%	8.50%	38.2%	10.0%	45.0%	1.50%	6.75%	
Loss of riparian habitats and floodplain connectivity (lakes)	5.0%	15.0%	4.25%	12.7%	5.0%	15.0%	0.75%	2.25%	
Invasive species (terrestrial)	1.0%	2.0%	0.85%	1.7%	1.0%	1.0%	0.15%	0.30%	

Note: 1, based on specific pressures listed below, taking into account potential overlaps. 2. Although baseline levels are not certain at an EU level, no additional restoration is expected to be required due to anticipated action under the WFD by Member States.

6.10.5 Key measures for restoration

The major initiative to achieve a Good Ecological Status of freshwater ecosystems in the EU is the establishment of River Basin Management Plans under the WFD. Effective conservation of freshwater resources requires integrated catchment management involving statutory controls, (agri-) environmental incentives and source targeted measures to reduce pressures from pollution and water abstraction. The WFD obliges Member States to take all necessary measures to reduce the impacts of pollution pressures and to restore freshwater ecosystems affected by pollution; to maintain minimum water flows, and to restore natural water flow regimes; to restore river connectivity for fish and other biological elements (see above); and to restore and maintain natural river channel dynamics as far as necessary to achieve Good Ecological Status or Good Ecological Potential.

As described above, we assume that the measures necessary to address most of the pressures on rivers and lakes will be carried out to maintain or restore to Good Ecological

Status or Good Ecological Potential under the WFD, and therefore are covered by Member States' existing and anticipated costs to 2020. It is also assumed that ecologically intact rivers and lakes generally do not require additional maintenance measures beyond those already required by the WFD to preserve their ecosystem structure and functions. The impacts of pressures such as excessive water abstraction and low flows on other ecosystems are considered in the inland marshes, forests and grasslands sections.

Some hydromorphological restoration measures will be required under the WFD in order to restore the water body to Good Ecological Status (ie the aquatic component of the ecosystem). But many of the large-scale measures go beyond Good Ecological Status needs and are primarily required to restore the ecosystem as a whole, including river banks, floodplains and lakes shores. The following section therefore only considers measures that are necessary to address pressures that are not covered by the WFD.

Key restoration measures for river riparian habitats and floodplain connectivity

The scale of river restoration measures varies enormously from small-scale measures to improve the habitat in short stream and river reaches, to measures restoring the connectivity between sections of watercourse, up to large-scale rehabilitation of river alleys and floodplains (Hansen, 1996). Measures to restore riverbank structure and habitat are part of projects at any scale.

Large-scale river restoration projects to reconnect rivers with their floodplains involve **de-embankment, widening of the river course,** and side channel restoration or construction to **re-connect freshwater habitats**, such as artificially created ox-bow lakes, cut-off meanders, disused gravel pits, and isolated wetlands. It is often necessary to **raise or lower the main river bed** in order to bring it in balance with the level of the floodplain, because the river bed may have been scoured out or silted up, and the floodplain may have sunk. These projects are associated with the re-creation or restoration of inland marshes and other wet habitats, which may include other measures to raise water tables (see inland marshes below for further discussion). **Restoration or creation of shallow water habitats** on the floodplain can be carried out at a small or large scale.

De-embankment and re-naturalisation projects will include measures to re-naturalise river banks, including removing over-shading by trees (where it has a negative effect) or planting trees and/or bushes where they are important for stabilising banks, controlling access by livestock to the river and/or preventing erosion at access sites by reinforcing the river bank and river bed.

There is a lack of data on the extent of hydromorphological restoration measures required to meet WFD targets, but in this study it is assumed that at least 10% of the degraded area along river banks will require each of the measures to restore them.

An increasingly widespread and expensive measure in river bank restoration is the **control of invasive plant species**.

Key restoration measures for lakes (riparian habitats and hydromorphological restoration only)

Key measures to restore lake habitats include measures to naturalise the bank structure and manage bank erosion (Perrow & Davy, 2002) including **removal of artificial bank linings**, **reprofiling lake shores and margins** to create shallow margins and refuge habitat for fish and zooplankton recruitment. When reprofiling lakes it is important to leave slopes that are gentle enough to allow the different types of vegetation to find an adequate niche during the flooded period and avoid erosion. Measures to restore / create habitats for particular species eg birds in lakes are usually carried out at the same time, including the creation of breeding islands and sandbanks protected from predators and human disturbance.

There is a lack of data on the extent of hydromorphological restoration measures required to meet WFD targets, but in this study it is assumed that at approximately 10% of the degraded area will require measures that go beyond WFD.

Table 6-43 Key maintenance and restoration and measures for rivers and lakes that are not covered by the WFD, and their required percentage coverage to address key pressures

The table refers to required percentage cover for either the total area, or the river or lake area separately, as specified in the measure. However for the costs calculation, the percentages are adjusted to account for 91% of the area being water bodies and 9% being rivers.

	Reconnect floodplain habitats, side channels	**De-embankment, raise or lower main river bed**	**Reprofile / restore river banks**	**Restore lake shores and habitats**	**Manage/ restore riparian vegetation to control invasives**
Loss of riparian habitats and floodplain connectivity (rivers)	10%	10%	10%		
Loss of natural dynamics and habitats (lakes)				10%	
Invasive species (terrestrial)					100% (riparian plants)

6.10.6 Costs of management and restoration measures not covered by WFD

Costs of river restoration measures not covered by WFD

The immediate issue when looking at costs of restoration in river projects is the scale of the work. River restoration ranges from work carried out by volunteers on a fishing stream to engineering works on the Danube, Europe's largest river. Clearly there is no European 'unit cost' for river restoration but LIFE project experience can provide some actual costs and can potentially compare similar projects in different countries (e.g. small upland rivers) on issues, solutions and costs. Each river will have its own set of circumstances, and the current study can only provide ranges of costs with some explanation. Some examples of costs of measures from LIFE projects are listed below, sourced by the Astrale group, and detailed in the Excel database accompanying this report. Costs are generally only quoted for the project as a whole or per length of river, but as the CLC land cover system only registers rivers at least 100 m wide, it is assumed here that costs per 100 m are equivalent to costs per ha of river area.

Key measures to control and eradicate invasive species in riparian habitats (rivers only)

Information on the costs of the removal of herbaceous invasive species along river banks appears to be scarce, but costs in one LIFE project in Italy averaged 440 €/ha (LIFE02 NAT/IT/008572 Toce River) . Removal of invasive trees cost 3,100 €/ha in another project in Austria (LIFE06 NAT/A/0126 Obere Drau II), but such measures are unlikely to be typical or widely needed. The cost of control of Japanese Knotweed (*Fallopia japonica*) in all infested riparian areas across the UK is estimated at 520 £/ha (650 €/ha) (Morling, 2008)[96]. This is judged to be a conservative estimate[97]. Japanese knotweed is one of the most widespread and damaging invasive herbaceous species in riparian areas but is difficult and expensive to control using current techniques. Himalayan Balsam (*Impatiens glandulifera*) control in a riparian meadow in the UK cost 47 £/ha (59 €/ha) (Wye Valley AONB, 2009). Control of herbaceous perennial IAS generally needs to be repeated for at least two years until all plants have gone.

In the absence of better information this study assumes an average cost of **500 €/ha** for managing invasive species along river banks.

Key restoration measures for river riparian habitats and floodplain connectivity

Member States are not obliged to carry out substantial hydromorphological restoration of heavily modified and artificial water bodies, provided they achieve Good Ecological Potential. If these measures are carried out in order to achieve wider habitat re-creation goals, they can be costly. Some examples of costs are given below.

LIFE project costs for **initiation / creation of a new main river course, channel re-shaping, and re-meandering** range from 5,700 to 211,600 €/100 m in Denmark, Germany and Belgium (LIFE05 NAT/DK/000153, LIFE05 NAT/DE/0057, LIFE04 NAT/DK/000022, LIFE07 ENV/B/0038). **Widening of the river course** by 10 to 20m along the Rhine cost 450,000 €/100 m (LIFE04 NAT/DE/000025). This study assumes an average cost of **100,000 €/ha** for re-naturalisation of heavily modified river courses.

Deconstruction of technical river banks (concrete, stone) and new river bank creation cost from 2,200 to 54,100 €/100m in Germany, Spain, and Austria (LIFE03 NAT/D/0006, LIFE05 NAT/E/000073, LIFE05 NAT/DE/0057, LIFE04 NAT/DE/000025, LIFE06 NAT/A/000127). Restoration of river banks (without deconstruction), incl. new stabilization cost from 1,105 to 67,700 €/ha in three different LIFE projects in France, UK and Italy (LIFE00 NAT/F/007277,

[96] The cost of control in riparian areas, where approximately 10,000 km of watercourse is estimated to be affected, is calculated at £52 million. This assumes both banks are affected to a width of 2 m from the edge of the water on both banks (Morling, 2008).
[97] http://www.jknotweed.com/news/controlling-japanese-knotweed-would-cost-uk-economy-2-billion

LIFE04 NAT/GB/000250, LIFE02 NAT/IT/008572). Re-vegetation of armouring rip-raps cost 31,600 €/ha in Spain (LIFE05 NAT/E/000073). Deconstruction of flood protection embankments cost 5,290 from to 45,000 €/ha in three different LIFE projects in Spain, Germany and Denmark (LIFE05 NAT/E/000073, LIFE05 NAT/DE/0057, LIFE00 NAT/DK/7116). It is clearly difficult to calculate an average, or typical, cost from such widely varying estimates, but for this we assume an average cost of **25,000 €/ha** for de-embankment and reconstruction of more natural river banks.

Reconnection of an artificially separated meander to the river cost from 25,000 to 34,800 €/100m in France, Germany and Greece (LIFE00 NAT/F/007277, LIFE04 NAT/DE/000025, LIFE 02 NAT/GR/8489). New **creation or revitalization (excavation) of an** artificially separated **meander**, oxbow lake, or gully cost from 6,900 to 586,000 €/ha in Germany and Spain (LIFE05 NAT/DE/0057, LIFE05 NAT/E/000073). Improvement or new **creation of shallow waters** in a floodplain cost from 1,200 to 262,000 €/ha in three different LIFE projects in Germany (LIFE03 NAT/D/0006, LIFE05 NAT/DE/0057, LIFE04 NAT/DE/000025). This study assumes an average cost of **1,500 €/ha** for reconnection of floodplain habitats, assuming that 15% of river lengths are on floodplains.

Costs of lake restoration measures not covered by WFD

Lake shore restoration by re-profiling shores, creating shallow undisturbed areas, islands and more natural land-water transition habitats, will cost several millions per water body as it may require major displacement of soil. An extreme example is the restoration plan to transform Lake Ijsselmeer in the Netherlands into a ‘future proof ecological system’, which has a projected total cost of around 850 million € for 1000 ha marshland, for a suite of measures including the creation of **large scale land-water zones (c. 4.6 million €/ha), creating foreshores** (c. 1.5 million €/ha) and the **construction of dams, breakwaters and deep pits to manage sludge** (c. 1.3 million €/ha) (Ecorys, 2011). On the other end of the spectrum the reported costs of restoring a single pond by sediment removal vary between 13,500 and 40,000 € per pond (RWS Waterdiens, 2012).

Within the scope of this study it was not possible to locate sufficient data to estimate an average cost of profiling lake shores etc and re-establishing riparian vegetation on them. **Therefore this report only estimates costs for the river ecosystem**. It is in any case likely that Member States will chose to restore 15% of degraded wetland ecosystems by carrying out less costly measures on other wetland types or less degraded lakes.

6.10.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total costs of achieving the 15% restoration target in river ecosystems. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 under the reference scenario, but in particular under the WFD (as described above). Maintenance costs are not calculated because it is assumed that maintenance requirements will be addressed through measures under the WFD. No re-creation cost is calculated because re-creation of this ecosystem is not required to meet the Target 2, or generally feasible in practice.

The estimate includes only river restoration measures that are not covered by the WFD and assumes hydrological and morphological measures will not be required for heavily modified or artificial water bodies. Due to a lack of data on combined measures, no estimate is provided based on overall degradation levels; therefore only one set of estimates is provided based on the proportion of river ecosystems that are impacted by key pressures and the costs of key measures that address them. See chapter 3 for a detailed account of the methodology and its assumptions.

The results provided in Table 6-44 indicate that the estimated annual costs (averaged over the 2010 – 2020 period), in addition to those expected under WFD measures, of achieving the **restoration target** in river ecosystems is between €4 and 18 million. However, it should be borne in mind that the true costs are likely to be higher because the CLC river land cover is a significant underestimate of the actual river area. As a result the figure presented should be treated as indicative only.

It is also important to note that the cost estimates only relate to rivers (due to inadequate cost data for lakes), and that under the CLC system the freshwater ecosystem is heavily dominated by the area of lakes. Therefore, the overall cost of maintaining and restoring the freshwater ecosystem is likely to be much higher than that estimated for rivers alone.

Table 6-44 Projected additional costs (€) in 2020 of achieving Target 2 within river ecosystems on the basis of current costs of key restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5) and some other costs described in section 3.4.4. The rivers ecosystem calculation assumes that all maintenance costs are covered by the WFD baseline. No overall habitat recreation costs are calculated, as the overall area of freshwater ecosystem has not decreased. Ecosystem area = 1,004,227 ha (rivers only).

Key Pressure	% Area requiring restoration in 2020		Key measure	Current one-off costs (€/ha) costs (€/ha)	% area applied to	Instru ment	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Reduced water levels	0.000%	0.000%	Reduce abstraction rates			WFD	-	-
Pollution	0.000%	0.000%	Dredging and biomanipulation			WFD	-	-
Modified water regimes	0.000%	0.000%	Restore natural river hydrolgy			WFD	-	-
Fragmentation of river connectivity	0.000%	0.000%	Remove barriers or add fish passes etc			WFD	-	-
Reduced river channel dynamics and diversity	0.000%	0.000%	Restore main river course, re-meander			WFD	-	-
Loss of riparian habitats and floodplain connectivity with rivers	1.50%	6.75%	Reconnect side channels etc	1,500	10%		226,000	1,020,000
			De-embankment, restore river banks	25,000	10%		3,770,000	16,900,000
Loss of riparian habitats and floodplain connectivity beside lakes	0.750%	2.25%	Restore lake shores and habitats	?	10%			
Invasive species (terrestrial)	0.150%	0.300%	Clear invasive species along banks	500	100%		75,300	151,000
RESTORATION TOTAL							***4,070,000***	***18,100,000***

6.11 COASTAL ECOSYSTEM – BEACHES, DUNES, SALTMARSHES, ESTUARIES & LAGOONS

6.11.1 Description of coastal ecosystems

The coastal ecosystem includes estuaries, coastal lagoons, intertidal areas, beaches, sand dunes, and coastal wetlands (ie saltmarshes and associated areas[98]). However this section does not consider rocky coasts and cliffs (see annex 1 for details).

Europe has a huge extent and diversity of coastal habitats ranging from the brackish shallow or rocky coastlines of the Baltic Sea, the North Sea coastline with its large intertidal flats, the oceanic Atlantic coastlines of Ireland, the UK, France and Portugal, to the high salinity Mediterranean Sea, and the brackish sandy Black Sea coastlines of Romania and Bulgaria[99]. Estuaries and lagoons are among the most productive ecosystems in the world, with both high ecological and economic values. European coastal ecosystems are highly modified by human impacts and infrastructure, and carry a high population density, major urban centres, and a large share of current social and economic development (European Commission, 2011c). Estuarine and coastal landscapes have been deeply modified and transformed. As an example, 15,000 km^2 of wetland, coastal lagoons, and tidal flats in the Wadden See region in the Netherlands have been embanked, drained and converted into arable land and pasture (Airoldi and Beck, 2007). The historical loss of coastal ecosystem biodiversity has been huge, and the rate of habitat loss continues to be high.

Coastlines and their ecosystems are inherently dynamic, being continually created, moved and destroyed by the action of the tides, wind, waves and rivers and their respective sediment loads. These coastal dynamics are highly reliant on long-range sediment movements and inputs from estuaries, river deltas, and coastal erosion, and therefore coastal restoration must be carried out with an integrated coastal planning approach at a sufficiently large scale (Pickaver and Ferreira, 2008). This section therefore considers the following coastal land cover types together:

Estuaries are river mouths within which the tide ebbs and flows, and which are dominated by salty or brackish water. Large estuaries are found mainly on the Atlantic coast. Estuaries deposit substantial amounts of sediment, creating mudflats and sand banks, sometimes with fringing vegetation or salt marshes. The CLC estuary area includes all the intertidal and salt-water dominated fringing habitats within the estuary area, except for large (> 25 ha) salt marsh areas.

Coastal lagoons are salty or brackish water bodies which are separated from the sea by a tongue of land or other similar topography. These water bodies can be connected to the sea at limited points, either permanently or for parts of the year only. Coastal lagoons are found around all European coasts, with a wide range of salinities from salt water to almost fresh water, and fringing marsh vegetation (see also the freshwater ecosystem section).

[98] It also includes salinas for commercial salt production (422), but salinas are not considered as natural ecosystems in need of restoration (although they may support similar specialised birds and invertebrates as natural lagoons) and are therefore neglected in this study.
[99] Six Member States have no coastal ecosystem as defined in this report: Austria, Czech Republic, Slovakia, Hungary, Luxembourg, and Malta (which has mainly rocky cliffs).

Intertidal flats are un-vegetated or sparsely vegetated expanses of mud, sand or rock lying between high and low tide water-marks. Sandy intertidal flats are integral components of coastal sand dune systems, and muddy intertidal flats are integral components of salt marsh; both provide sediment flows. Intertidal flats that are inside large estuaries are included in the estuaries area (see annex 1 for more detail on the definition). Intertidal habitat is found only on North Sea coastlines (Germany, the Netherlands, Denmark, Belgium, the UK and France) and Atlantic coastlines (Ireland, the UK, France, Spain and Portugal). This land cover is not present in the Baltic Sea, the Black Sea, or the Mediterranean Sea because of their small to tiny tidal ranges.

Beaches and dunes include coastal sand dunes, sand or shingle above the high water mark. Vegetated shingle banks and ridges are a particularly highly threatened coastal habitat. Intact dune systems consist of a dynamic series of successional habitats from the beach to the inland climax vegetation, including:

- Beach high tide strandline with annual vegetation
- Sparsely vegetated shifting dunes and white dunes with *Ammophila arenaria*
- 'grey' dunes that are more or less permanently stabilised by vegetation
- Dunes with sparse dune scrub such as sea buckthorn (*Hippophae rhamnoides*) or *Salix* or in the Mediterranean with pseudo-steppe grassland or *Euphorbia terracina* or *Crucianella maritime* scrub
- Dune slacks and other dune wetlands
- Dune grassland in dry interdunal depressions (in Mediterranean)

Machair, a coastal sand-plain grassland, used for grazing and cutting forage and thatch, only found in Scotland (UK) and Ireland, is also included in this category.

This study assumes that of the total beach and dune area (324,693 ha), 31% is beach (with at least 11% shingle), 15% is embryonic and shifting dunes, 23% is vegetated dunes, 6% is dune slacks and other wet dune habitat, and 25% is dune grassland (plain) or machair[100].

Older dunes or shingles covered in large expanses of heath, scrub or woodland may be classified as heath or sclerophyllous scrub or forest land cover (see annex 1 for more information). Likewise, dunes afforested with large plantations may be classified as forest.

Salt marshes are vegetated low-lying areas on the coast susceptible to flooding by sea water. They are often in the process of filling in, gradually being colonised by halophilic plants. They can consist of vegetated flats, creeks, pools and saline reed beds. Smaller saltmarshes (<25 ha) within estuaries are included in the estuaries area (see annex 1 for more detail on the definition).

Salt marsh habitat can be divided into three zones, each of which have different characteristic vegetation types and fauna, and which are subject to different pressures.

- low marsh occurs between the mean high water level of neap tides and mean high water. The vegetation is sparse with a limited number of species, such as *Salicornia*,

[100] This is based on the proportions of these habitats in the reported Annex I habitats; see annex 1 for details.

Spartina, and *Aster* spp. The sediment is very dynamic, with waves of accretion and erosion.

- middle marsh occurs between mean high water and the mean high water level of spring tides. The vegetation is penetrated by an intricate network of creeks and pools which fill with salt water at high tide.
- high marsh occurs above mean high water level of spring tides. The natural vegetation is very dense and diverse with many flowering plants, dominated by sea purslane *Atriplex purtulacoides* and sea couch *Elytrigia atherica* or *Phragmites*. However most surviving areas of high marsh have been grazed and this alters the vegetation to lower-growing grasses (Ausden, 2007).

6.11.2 Extent of coastal ecosystems

The areas of coastal ecosystems in each Member State are provided in Table 6-45. The CLC data gives a total of around 2.5 million ha of coastal ecosystem (not including rocky or artificial coasts) (Airoldi & Beck, 2007). Almost half (45%) of this coastal ecosystem is intertidal flats, 31% is estuaries and coastal lagoons, 12% is beach and dune systems, and 12% is saltmarsh (more details are available in annex 1). It is not possible to assess how much of the estuary area is saltmarsh, beach or dune, intertidal zone, or open water, so the areas for these habitats will be underestimated. It should also be noted that, generally, Member States' data on the extent of their coastal habitats and habitat loss or change is very limited (Normander et al, 2009; UK NEA, 2011b). For example, for some UK coastal habitats estimates of their total area vary by up to 50% (UK NEA, 2011b).

This study uses the CLC area data for beaches and dunes of 324,693 ha, and for saltmarshes of 297,982 ha.

The table shows, for comparison, the proportion of rocky coastline (length) in each Member State. Rocky coastlines are not considered in this section because of lack of CLC data on their extent (see annex 1 for details). It is important to note, however, that hard and soft rock cliffs are integral parts of the coastal ecosystem both as ecological habitats in their own right and as sources of sediment for other coastline habitats. In some Member States, including Finland, Greece, Ireland, Malta, Portugal, Slovenia, Spain, Sweden, and the UK, rocky cliffs are the predominant coastline form.

Table 6-45 Areas of coastal ecosystems CLC (hectares)

The columns shaded in grey denote the area data that were used in this study. Data relate to 2006 CLC data except where indicated as * where data are for 2000. The data for Greece are subject to large inaccuracies. Data for coastline length and proportion of rocky and artificial coastline in 2001 are taken from (EUROSION, 2004a). Coastal length is calculated from cartography at scale 1:100,000 using a total coastline length of 100,925 km. NB As coastlines are fractal, estimates at different scales vary dramatically.

Land cover (ha)	Beaches and dunes CLC 331	Salt marshes CLC 421	Intertidal flats CLC 423	Coastal lagoons CLC 521	Estuaries CLC 522	Total coastal CLC	Share of EU27 coastline	Rocky coast proportion	Artificial coast proportion
Austria						-	-	-	-
Belgium	1,394	458	2,126		4,258	8,236	0.10%	0%	34%
Bulgaria	2,220	32		365		2,617	0.35%	n/a	n/a
Cyprus	5,138	1,964				7,102	0.07%[101]	9%	20%
Czech Republic						-	-	-	-
Denmark	9,022	30,377	58,814	46,581		144,794	4.56%[102]	9%	12%
Estonia	4,358	392		1,584		6,334	2.52%	6%	1%
Finland	1,194	13,360			69	14,623	13.89%[103]	57%	1%
France	40,627	80,129	212,762	76,258	53,646	463,422	8.17%[104]	40%	15%
Germany	13,185	18,028	284,593	108,209	21,292	445,307	3.49%	5%	18%
*Greece**	*48*	*7*				*55*	13.65%	50%	4%
Hungary						-	-	-	-
Ireland	10,888	4,848	52,821	817	33,432	102,806	4.54%	57%	3%
Italy	78,778	41,329		96,729	252	217,088	7.40%	43%	8%
Latvia	4,595	73				4,668	0.53%	0%	3%
Lithuania	2,622			40,269		42,891	0.26%	3%	12%
Luxembourg						-	-	-	-
Malta						-	0.17%	88%	7%

[101] only 20% of Cyprus is reported here (ie excludes Turkish Cyprus)
[102] Excludes Greenland and Faroes
[103] Excludes the Åland Islands
[104] Includes French Guyana, Guadeloupe, Martinique and Reunion (but excludes the Territoires Outre-Mer)

Netherlands	15,160	9,369	228,189		22,785	275,503	1.26%[105]	0%	60%
Poland	5,642			44,334		49,976	0.63%	0%	3%
Portugal	11,933	18,459	1,989	8,550	44,994	85,925	1.18%[106]	51%	5%
Romania	17,286	5,482		70,341		93,109	0.35%	N/A	N/A
Slovakia						-	-	-	-
Slovenia	614	219				833	0.05%	53%	18%
Spain	47,611	29,475	6,859	21,004	12,405	117,354	6.52%[107]	61%	10%
Sweden	4,856	1,779		6,778	34,122	47,535	13.44%	56%	1%
United Kingdom	47,522	42,202	275,690	674	27,546	393,634	17.22%[108]	60%	5%
TOTAL	**324,693**	**297,982**	**1,123,843**	**522,493**	**254,801**	**2,523,812**		**47%**	**6.4%**

[105] Excludes Aruba and the Netherlands Antilles. The Dutch coastline has been significantly reduced by the closing of the Zuiderzee by the Afsluitdijk in 1932 (250 km) and the closing of several sea arms by the Delta Works (700 km).
[106] Includes the Azores and Madeira
[107] Includes the Canary Islands but excludes Ceuta and Melilla
[108] Excludes the Isle of Man, Channel Islands, Gibraltar and UK OCTs

6.11.3 Key pressures on coastal ecosystems – 2010 baseline levels

This section attempts to estimate the overall levels of degradation of the ecosystem type and specific pressures at the baseline year, which for this study is taken to be 2010. See Chapter 2 for a discussion of the estimation of baseline levels and available data.

Coastal ecosystems are suffering from degradation throughout Europe. The EEA in 1999 considered that less than 15% of the European coastline is in 'good' condition (quoted in (Airoldi & Beck, 2007). Notably however, a fifth of the Annex I coastal habitat area in the EU-25 was reported as having an unknown status, plus the assessment did not include the coasts of Romania and Bulgaria (EEA, 2010a). In general, reliable data on the extent of coastal habitats and the magnitude of pressures they face is very limited (Normander et al, 2009; UK NEA, 2011b). Degradation is quantified for all coastal ecosystems as far as possible, but it was only possible to carry out the full analysis of restoration and maintenance costs for beaches and dunes and for saltmarsh.

Ecosystem loss

According to CLC data the EU has lost **0.2%** of its dunes and salt marshes between 2000 and 2006[109], and gained 0.2% to the area of intertidal mudflats, coastal lagoons and estuaries (EEA, 2010a). Over the last decade, around two-thirds of the loss of dunes and salt marshes was due to coastal erosion, a fifth due to the spread of economic sites and infrastructures, and smaller amounts to urban residential development, afforestation, creation of water bodies, and conversion to agriculture (EEA, 2010a).

Artificial surfaces increased on Europe's coast by **0.3%** between 2000 and 2006 (EEA, 2010a). The extent of artificial surfaces[110] along EU coastlines is shown in Table 6-45. Belgium has over a third, and the Netherlands almost two-thirds of their coastline as artificial surfaces; seven other Member States have 10-20% (EUROSION, 2004a). The highest increase in artificial surfaces (20-35%) has been in the coastal zones of Spain, Ireland and Portugal. About two thirds of the Mediterranean coastline is reported as dominated by artificial surfaces (Airoldi & Beck, 2007). The intertidal habitat is highly fragmented by breakwaters, seawalls, jetties and pilings, which disrupt natural intertidal sediment patterns and species dispersal, and provide substrates for the colonisation of exotic species (Bulleri and Chapman, 2010). Salt marsh continues to be lost to industrial infrastructures associated with port developments, coastal energy infrastructures such as power stations, urbanisation, and tourist developments, with particularly high pressure on the Mediterranean (EEA, 2010f). Salt marshes are also used as waste tipping and land fill sites. A significant area of grey dunes has been lost to tourism infrastructure, golf courses and residential development, and existing sites are often fragmented by infrastructure and impacted by coast defence works (Houston, 2008a).

[109] It is important to note that this area is not directly comparable to the total CLC land cover area data for 2006, as the land cover change is calculated from land cover data at a resolution of 100m, whereas the total area in 2006 is calculated at a resolution of 250m.

[110] Including seawalls, dykes, quays, embankments, harbour infrastructure, man-made protective rocky strands, but not sand strands

Overall, **at least 11%** of Europe's coasts are now actively retreating due to coastal erosion[111], some of this despite coastal protection works, and another 4% is artificially stabilised (Airoldi & Beck, 2007; Salman et al, 2004). According to the EUROSION project, **22%** of the EU beach coastline and **13%** of the muddy coast is eroding (EUROSION, 2004a). In 2001 (EUROSION, 2004a), more than 80% of the Slovenian coastline was impacted by coastal erosion; in Belgium, Poland, Romania and Bulgaria more than half of the coastline was impacted by coastal erosion; in Latvia, Germany, France, UK, Portugal, Italy, Greece, and Cyprus, a third or more of the coastline was impacted by coastal erosion. In contrast, in Finland, Sweden and Estonia the coastlines are accreting.

Increases in coastal erosion are caused by disruptions in the supply of sediment from along the coast and from rivers. Because of dams on major rivers, huge volumes of sediment are no longer reaching estuaries to replenish beaches and mudflats, as illustrated in the estuaries and beaches of the Ebro, Urumea, Rhone and Duoro (Salman et al, 2004). Some Iberian and Mediterranean rivers transport less than 10% of the annual volume of sediment they discharged in 1950 (EEA, 2006b); the Ebro transports less than 1% of the sediment it transported a century ago (Tena et al, 2011). Coastal defence engineering combined with a disrupted sediment supply causes profound alterations to intertidal flats and to longshore sediment transport, which is likely to have unknown knock-on impacts for downdrift coastal habitats and for coastal marine biodiversity (Airoldi & Beck, 2007). For example, the large dykes built in front of Belgian and Netherlands dune systems led to the disappearance of the dry beach section on many beaches. However, coastal erosion results from the cumulative impact of a wide range of natural and human-induced factors, none of which may be considered as the single cause for erosion, so it is difficult to quantify the impact of different factors.

Coastal erosion is basically a natural process that is necessary to maintain the natural coastal dynamics that shape coastal habitats and maintain their biodiversity. Erosion only results in overall habitat loss if new habitat can no longer be created at the same rate as it is lost. Land reclamation for agriculture or urban use means that most coastal habitats are now blocked off by embankments or walls and can no longer migrate inland ('coastal squeeze'). At least 10% of the EU coastline has been engineered with coastal infrastructure, and **more than 6% has 'hard' engineering structures** – seawalls and embankments, detached breakwaters, or revetments (EEA, 2006b)[112].

Erosion is one of the principal threats to intertidal mudflats and saltmarshes, and to some sand dune systems. Salt marshes are created from the accretion of sediments from intertidal mudflats in continual cycles of sediment deposition during normal low energy conditions and erosion during high tides and storms (Nottage and Robertson, 2005). Salt marsh can only be maintained in a static location if the balance between creation and accretion is stable. The larger scale dynamics of the coastline or estuary determine whether the overall result is that the saltmarsh expands into the sea as the intertidal mudflat

[111] As of 2001, 20,000 km^2 was eroding (Salman et al, 2004) and the EU has around 143,263 km^2 of coastline (Airoldi & Beck, 2007).

[112] For example, Belgium has only 65 km of shoreline, but less than half is protected by natural habitats - beaches and dunes. More than half (35 km) is protected by seawalls in combination with beaches and 3 km by breakwaters (coastal harbours).

expands, or that the saltmarsh retreats inland as the intertidal mudflat retreats. However, ‘coastal squeeze’ means that many saltmarshes are eroding away or have completely disappeared (Nottage & Robertson, 2005). For example, England and Wales are suffering a net loss of around 50 ha (**0.12%**) of saltmarsh a year to erosion (Environment Agency, 2011). High marsh has been disproportionately lost - for example, in the UK high marsh habitat has almost disappeared (UK NEA, 2011b) - and in general, high marsh is much more species rich than low marsh. In contrast, saltmarsh in Ireland is not considered to be much affected by coastal erosion (McCorry and Ryle, 2009).

Another significant cause of the loss of dunes in all Member States has been afforestation, especially in France, Denmark and the UK (Houston, 2008b). In Wales (UK), for example, plantations cover approximately 21% (1700 ha) of the dune resource, and of this, a large proportion would have included grey dunes (JNCC, 2007). Another source specifies that afforestation affects approximately 14% of the total dune area of England, Scotland and Wales (UK)[113]. Along much of the French and south Baltic coastlines the remaining strip of dune habitat is “trapped” between coastal defences and the afforested fixed dunes (Houston, 2008b). Afforestation rates in recent decades have been very low due to improved protection and poor profitability of timber production in such ecosystems, and they are likely to decline to negligible levels on a European scale by 2020.

This study assumes that the annual rate of loss of sand dune and beach ecosystems from 2000 to 2010 was between **0.02% and 0.1%**. Saltmarsh is also considered to have been lost at the same rate.

Key pressures on estuaries and coastal lagoons - baseline

Overall degradation of estuaries and coastal lagoons
Typically, estuaries and lagoons are strategic locations for major port developments and industrial developments, and marinas. The associated maintenance and development activities pose significant threats to these ecosystems. Land reclamation for port or industrial developments or urban sprawl results in habitat loss, as does shoreline reinforcement for flood defence. Member States reported 100% of their Annex I estuary habitat as being in unfavourable-bad status in 2007 (ETC/BD, 2008a). The Annex I estuary habitat area is much larger than the CLC estuary area, so it can be assumed that **95 to 100%** of the estuary area is degraded. Member States reported 100% of their Annex I coastal lagoon habitat as being in unfavourable-bad status in 2007 (ETC/BD, 2008a). The Annex I habitat is equivalent to 77% of the CLC coastal lagoon area, but assuming that this represents the most ecologically intact area, it can be assumed that **95 to 100%** of coastal lagoons are degraded.

Changed salinity and hydrographical functioning due to modifications in the connections between sea and freshwater inflows, and to drainage and over-abstraction of groundwater
Coastal lagoons (and estuaries to a lesser degree) differ in their salinity and hydrological and sediment dynamics from the sea because they are partially separated from the open sea and are subject to the influence of river water. Many coastal lagoons have been drastically

[113] http://www.coastalwiki.org/coastalwiki/Sand_dune_-_Country_Report,_Great_Britain#Afforestation

altered due to being opened up to the open sea so that they can be used for recreational or commercial shipping. This is especially widespread in Mediterranean coastal lagoons which were often only separated from the sea by narrow sand bars or spits, and are now inundated by highly saline Mediterranean seawater. For example, the French Mediterranean Étang de Leucate (Salses) has been profoundly modified by the influx of saltwater resulting from an enlarged opening to the sea for a tourist marina development, which sharply increased salinity. The reverse effect is created by river engineering or infrastructure that drastically increases the inflow of freshwater into coastal lagoons. For example, the French Mediterranean Étang de Berre suffered a dramatic drop in salinity following the construction of a hydro-electric power station that discharges piped water from the Durance river into the lagoon each year, equal to three or four times its volume. Other coastal lagoons are threatened by lowering water tables due to drainage and over-abstraction of groundwater for irrigated agriculture (Pérez-Ruzafa et al, 2011). The Mar Menor coastal lagoon and wetlands in SE Spain are currently expanding due to drainage from irrigated agriculture, and this has altered the bird communities (Robledano et al, 2010). A few coastal lagoons are fed by karstic aquifers, and these are highly sensitive to saline intrusion.

Despite the importance of this pressure quantitative data are lacking on the proportion of lagoons affected across the EU.

Erosion and altered sediment dynamics due to dredging and extraction of sand and gravel
Dredging operations to maintain or improve port access or to extract sand or gravel can significantly affect the hydrological functioning of estuaries and lagoons, resulting in increased erosion and the loss of sand banks, mud-flats and salt-marshes that rely on sediment accretion. In the UK, it is estimated that about 25% of estuarine intertidal flats have been removed, with some estuaries such as the Tees having lost up to 80% (UK Biodiversity Group and OSPAR Commission quoted in (Airoldi & Beck, 2007). According to information on the proportion of estuarine water bodies that Member States have declared as heavily modified under the Water Framework Directive, at least 50% of the CLC estuary area is heavily modified. However, there is insufficient information available to (see annex 1 for details of the assumption).

Succession of vegetation in coastal lagoons due to lack of ongoing management
Coastal lagoons are invaded by fringing vegetation, such as reeds, sedges, or scrub, unless the surrounding habitats are maintained through grazing and/or cutting. Coastal lagoons are also susceptible to invasion by algae and other floating vegetation under the influence of eutrophication (see Chapter 4 for more details). Information is lacking on the extent of these pressures, but it is likely that most lagoons outside protected areas are under-managed, as well as some within them. On this basis we assume that **30-50%** of the coastal lagoon area requires management of bank-side and floating vegetation.

Invasive species in estuaries and coastal lagoons
Invasive species are having a serious impact on coastal ecosystems in the EU, and often become established more easily in ecosystems that are already degraded by other pressures. Although numerous case examples exist [some to be added] of ecosystem

degradation by alien invasive species, insufficient information is available to quantify overall impacts at an EU level.

Eutrophication from sewage and other water pollution – toxic algal blooms and anoxia
Eutrophication - an overload of nitrogen, phosphorus, and organic matter from agriculture, sewage, industrial effluents, and aquaculture – is one of the most important problems facing European coastal waters (EEA, 2010a); see section 5.3 for more detail. Eutrophication has increased the turbidity of water and reduced the oxygen content of European coastal waters, and led to recurring algal blooms[114] and anoxic conditions that pose a serious threat to estuarine and lagoon communities. Toxic algal blooms (eg *Dinophysis*) cause closures of fisheries and mussel beds. Eutrophication is a major problem in the Baltic Sea, which has changed from nutrient-poor, clear water to turbid eutrophic water, and in German, Belgian, French and Scottish coastal waters (Conley et al, 2011). For example, the Odra lagoon in Poland is highly eutrophied, with a low water transparency often below 1 m, and heavy and periodic cyano-bacteria blooms in summer; there is a gradual improvement due to the cleaner Oder river, but the water continues to be influenced by pollutant deposits in the sediments. In the Mediterranean, eutrophication is a common problem in sheltered coastal waters and coastal lagoons near urban areas (EEA, 2007). The Danube Delta and the Black Sea coast affected by the delta carry a high nitrogen and phosphorus load, which has caused severe eutrophication problems and anoxia, with death of fish and mussel beds. At source measures to reduce eutrophication are discussed in section 5.2 and 5.3.

Water pollution from chemical contaminants
Chemical contaminants from agriculture, industrial effluents, mining, oil spills, and anti-fouling chemicals from ship hulls (including heavy metals, pesticides, biocides, PCBs, dioxins and hydrocarbons) enter estuaries and coastal areas and accumulate in sediments, and marine pollution is a major concern (Airoldi & Beck, 2007; EEA, 2007). Discharges have been decreasing in recent years (EEA, 2007), but concentrations in the Baltic, Mediterranean and Black Sea estuaries which have little turnover with ocean water will remain high for a long time (Tockner et al, 2009). Chemical pollution needs to be primarily dealt with by reducing sources.

Most of Europe's estuaries and coastal lagoons are affected by a combination of most or all of these factors, and it is very difficult to quantify the degree of degradation from each pressure. Furthermore, the amount of information available on the state of estuaries and coastal lagoons is very poor, as shown in the case of Ireland (National Parks and Wildlife Service, 2008).

Other impacts on the near shore and marine environment, such as overfishing and the destruction of seabed habitats from trawling, are discussed in section 6.12.

Key pressures on dunes and beaches - baseline

Overall degradation of dunes and beaches

[114] Eg the green alga *Valonia aegagropila*

According to the Article 17 reporting around 80% of the area of Annex I beach and dune habitats (see annex 1 to this report for details) was in unfavourable condition in the EU-25 in 2007. The monitoring of these Annex I habitat types is generally good with no major gaps in coverage and there is a reasonable overlap between the dune ecosystem area (324,693 ha) and Annex I habitat area (344,764 ha). Therefore the Article 17 assessments provide a reasonably representative and reliable indication of the overall level of degradation of the ecosystem type. Consequently, with some rounding, overall degradation levels in 2007 are assumed in this study to be **minimum 60% and maximum 80%** of the beach and dune ecosystem area.

Loss of natural coastal dynamics and hydrology due to coastal defences, embankments, and land reclamation

Dune systems form a dynamic system that is continually fed by sand from intertidal sand flats, and subject to periodic erosion ('blow-outs') from wind and waves, which form new dunes and scour out new dune slacks. The larger scale dynamics of the coastline or estuary determine whether the overall result is that the dune system expands into the sea or wanders inland. Nowadays most dune systems have been stabilised by physical structures and/or by planted vegetation and are no longer able to follow their natural dynamic (Provoost et al, 2011). Along most of the Netherlands and Danish coasts the dune systems are separated from the sea behind coastal engineering works or a high sand dyke. Dune systems across Europe are suffering from a shortage of incoming sand (Houston, 2008b). This study assumes that **at least 50% and maximum 75%** of Europe's beaches and dunes are degraded due to a lack of natural coastal dynamics.

Succession and loss of heterogeneity through lack of appropriate management

The open dune landscape in Europe has historically been maintained through grazing by rabbits and domestic livestock, and by cutting scrub for firewood (Houston, 2008b). Overall, this study assumes that theoretically, a maximum of 50% of the beach and dune area (ie the fixed dunes, dune grassland, and dune wetland) needs management to hold back scrub development. Overgrazing has been a problem on dune systems in the past, causing excessive erosion, but is no longer an issue in most places. Decreased grazing pressure, combined with eutrophication and lack of coastal dynamics, is leading to species loss through succession to scrub and woodland, loss of bare sand patches, and loss of heterogeneity at the local and landscape scales (Houston, 2008b). Dune slacks, for example, shift from pioneer to more mature stages after 20 to 30 years, although management can delay succession on some sites (Grootjans et al, 2002). The crash of rabbit populations in north-west Europe has had a profound effect on the small-scale habitat heterogeneity of dune vegetation. In the UK, it is estimated that dune systems would need to have 10 to 30% open dune habitat in order to conserve their species mix, whereas currently only 2% is open (Howe et al, 2010).

Given that 50% of the ecosystem needs management and that most is unmanaged outside protected areas, it seems reasonable to assume that between **10% and 40%** of the sand dune area is degrading through succession and loss of open habitats, because of lack of grazing and scrub management.

Degradation of dune slacks and other dune wetlands due to hydrological modification and lack of natural dynamics
Dune wetlands are particularly species rich and can host rare plant species, and are particularly threatened habitats (ETC/BD, 2008a). Their continued existence is dependent on both dune dynamics and the interplay between seawater and the freshwater dune aquifer that is supplied by rainfall. Lack of natural dynamics poses a particular threat to dune slacks as it inhibits their formation and interferes with their characteristic hydrological fluctuation between seawater and groundwater that delays succession (Grootjans et al, 2002; Houston, 2008a). Dune slacks in the Netherlands are highly threatened as a result of drinking water abstraction that has lowered water tables (Grootjans et al, 2002), although in some dune systems the water table has spectacularly risen after abstraction ceased. Dune slack habitat was reported under the Habitats Directive in the EU-25 on 19,975 ha, of which 93% was in unfavourable condition (and 6% in unknown condition) (ETC/BD, 2008a). This study assumes that between **5% and 10%** of the dune area is dune wetland, and that all of this is degraded due to modifications in the hydrology of the dune system and lack of pioneer habitats.

Invasive species in dune systems
Invasive species are cited by Member States as one of the major pressures on dune systems (ETC/BD, 2008a). Impacts and measures to deal with invasive alien species are discussed in section 2.4.

Eutrophication from nitrogen deposition
Sand dune vegetation is particularly sensitive to nitrogen deposition (Dise, 2011). Eutrophication on dunes leads to grass and moss encroachment and the spread of ruderal species, and the loss of calciphilous species in dry dunes (Houston, 2008b). A considerable amount of the management activity in dune systems can be attributed to the effect of eutrophication (Houston, 2008b). See section 5.5 for more detail on pressures and measures related to nitrogen deposition.

Key pressures on salt marshes and intertidal flats - baseline

Overall degradation of salt marsh
According to the Article 17 reporting of Annex I saltmarsh habitats (see annex 1), almost all the reported 353,535 ha is in unfavourable conservation status, and the rest has unknown status (ETC/BD, 2008a). This is a larger area than the CLC saltmarsh area, so it is assumed that **95 to 100%** of the saltmarsh area is degraded in some way.

Separation from the sea and impoundment
Saltmarsh has been reclaimed from the sea through the use of embankments and sluices and used as grazing marsh for centuries. Some of this saltmarsh area now no longer has any traces of salt water influence so that it is effectively lost as saltmarsh habitat and has become pasture, although it may still offer valuable habitat to birds and freshwater flora and fauna in the dykes. The challenge is how to identify degraded saltmarsh that is still intact enough to be restored. In the Severn Estuary, for example, there are a total of about 1400 ha of intact marshes and about 84,000 ha of enclosed marshes, and this ratio of 14% intact to 86% enclosed marshes may be common to other estuaries along the Atlantic and North Sea coasts (Allen 2000 quoted in (Airoldi & Beck, 2007)). Some saltmarsh has been re-

created as the result of past embankment breaches; for example, in the UK and the Netherlands, more than 2007 ha and 2590 ha respectively have been reinstated since 1870 (Wolters et al, 2005), but these areas are very small in comparison to the impounded saltmarsh. For this study it is assumed that **at least 50% and maximum 80%** of saltmarsh is being degraded due to separation from the sea and impoundment.

Inappropriate grazing – overgrazing, undergrazing
Introduction of grazing by domestic livestock on formerly ungrazed or lightly grazed salt marshes can have a deleterious effect, for example on Atlantic salt meadows (Doody, 2008). Heavy grazing pressure, for example by sheep, can severely reduce species diversity (Ausden, 2007). Conversely, reduction or cessation of grazing on historically heavily grazed salt marshes and coastal meadows results in a dense overgrown, species-poor sward unsuitable for typical bird species (Ausden, 2007). Overgrazing is considered to be the most important pressure degrading saltmarsh in Ireland, with an estimate that around 15% is damaged by overgrazing (McCorry & Ryle, 2009). For this study it is assumed that **at least 50% and maximum 80%** of salt marsh is degraded due to inappropriate grazing (undergrazing or overgrazing).

Invasive species on salt marsh
See section 2.4 for discussion of the impacts of invasive alien species and costs of measures to control and eradicate them.

6.11.4 Key pressures and extent of degradation in 2020

Ecosystem loss to 2020
It is expected that losses of coastal ecosystem to economic development (ports, tourism, energy & transport infrastructure) will continue to 2020. Shipping traffic in pan-European seas is likely to continue to increase rapidly in the next decade (EEA, 2007). Energy exploitation and generation may have a profound impact on coastal habitats in the next decade, such as tidal energy in estuaries.

Climate change will further increase the pressures on Europe's coastlines by both increasing the frequency and intensity of storms, waves and wind, and by raising sea levels. The Global Vulnerability Assessment has estimated that the loss of European coastal wetlands due to sea level rise would exceed 4,500 km² by 2020, which corresponds to a loss of 51% of the 1990 area (Salman et al, 2004). Low-lying coastal areas subject to strong tidal regimes and storm surges are most vulnerable. Along the Mediterranean coast, climate change will increase sea surface temperatures, sea acidification, and desertification of coastal habitats (EEA, 2007). Coastal wetland losses along the Mediterranean coast and the Baltic coast could be very high. Threats and costs of measures related to climate change are not dealt with directly in this chapter, but can be considered to be an escalation of the kinds of restoration costs described here.

For the purpose of this study, it is assumed that the coastal ecosystem will continue to be under pressure from development, combined with coastal erosion. Therefore loss of beach and dune habitat will continue at minimum **0.02**% and maximum **0.15%** annually between

2010 and 2020. Saltmarsh habitat will continue to be lost at minimum **0.02%** and maximum **0.15%** annually between 2010 and 2020.

Possibilities for ecosystem creation to 2020
Due to the escalating costs of coastal defence, many European Member States have adopted a policy of 'managed realignment', meaning that they are deliberately abandoning or modifying stretches of sea defences and allowing the coastal habitats to expand inland. This has created a significant potential for coastal habitat creation from low grade agricultural land. For example, the UK has a target to create 3,600 ha of new saltmarsh and mudflat by 2015 (adding 9% of existing area). Coastal habitat restoration and re-creation is considered to be the best way of adapting Europe's coastlines to the impacts of climate change. See conclusion for further discussion.

Key degradation pressures to 2020

Habitat degradation due to lack of coastal dynamics
Coastal habitats will continue to degrade over the next decade unless large-scale interventions that restore coastal dynamics can be initiated. For example, most sand dune systems are currently fossilised, and are rapidly losing their remaining open habitat areas (Provoost et al, 2011). It is likely that unless large-scale remobilisation of dune systems is undertaken, half of the remaining pioneer habitats will have been lost by 2020 (Grootjans et al, 2002). It is therefore assumed that the area of beach and dune degraded by lack of coastal dynamics will increase to at least **55%** and a maximum **80%** by 2020. It is assumed that, in the absence of active restoration and recreation measures, the ratio of impounded to naturally dynamic saltmarsh will continue to be similar as at present.

Saline intrusion in coastal habitats
Saline intrusion represents one of the greatest increasing threats to coastal habitats, especially in the Mediterranean area, through a combination of over-abstraction of groundwater reserves and longer drought periods, rising sea levels and more frequent storm surges. It can therefore be assumed that coastal lagoons will still be almost **100%** degraded in 2020, at least to a certain extent.

Grazing and mowing management of saltmarsh and sand dune grassland
The future of extensive grazing and mowing management is dependent on the future of agri-environment schemes for these habitats, as few managed areas are sustained through production revenues such as the marketing of premium saltmarsh lamb. It is difficult to predict the future of these schemes, so this study assumes that management needs will continue on more or less the same level to 2020.

Afforestation of dunes
It is assumed that afforestation of dunes will no longer be carried out in the next decade, but unless active restoration activities are undertaken, the proportion of afforested dunes will remain the same as at present, ie at least 5% and maximum 15% of the area.

6.11.5 Extent of degradation

Table 6-46 and Table 6-47provides a summary of the estimated level of overall degradation and degradation resulting from each key pressure on coastal sand dunes and beaches, and saltmarshes respectively. Insufficient data are available to quantify pressures on other coastal ecosystem types and therefore maintenance and restoration costs for these ecosystems are not estimated in this study.

Table 6-46 Estimated baseline and expected 2020 percentage degradation levels and conversion rates, and corresponding requirements to achieve a 15% restoration target for coastal sand dunes and beaches

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5. % restoration required = expected degradation level minus the maximum level to achieve target.

a) Annual degradation rates resulting from each key pressure requiring maintenance measures

Degradation source / key pressures	Degradation rate 2010		Expected in 2020	
	Min %/y	Max %/y	Min %/y	Max %/y
Overall	**0.500%**	**2.000%**	**0.500%**	**2.000%**
Lack of appropriate vegetation management	0.500%	2.000%	0.500%	2.000%

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels		Maximum level for 15% restoration		Expected in 2020		% restoration required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Overall	**60.0%**	**80.0%**	**51.0%**	**68.0%**	**65.0%**	**90.0%**	**14.0%**	**22.0%**
Lack of natural coastal dynamics	50.0%	75.0%	42.5%	63.7%	55.0%	80.0%	12.5%	16.2%
Lack of appropriate vegetation management	10.0%	40.0%	8.50%	34.0%	10.0%	40.0%	1.50%	6.00%
Altered hydrology & salinity	5.00%	10.0%	4.25%	8.50%	5.0%	15.0%	0.750%	6.50%

c) Conversion rates

Conversion causes	Conversion per year 2000-2010		Conversion per year 2010-2020		Total converted over 2000-2020		% re-creation required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Afforestation	0.010%	0.100%	0.010%	0.100%	0.200%	2.000%	0.030%	0.300%
Total	**0.010%**	**0.100%**	**0.010%**	**0.100%**	**0.200%**	**2.000%**	**0.030%**	**0.300%**

Table 6-47 Estimated baseline and expected 2020 percentage degradation levels and conversion rates, and corresponding requirements to achieve a 15% restoration target for saltmarshes

See text for derivation of estimates and information sources. Pressures do not include wide-scale pollution pressures, which are discussed in section 5. % restoration required = expected degradation level minus the maximum level to achieve target.

a) Annual degradation rates resulting from each key pressure requiring maintenance measures

Degradation source / key pressures	Degradation rate 2010		Expected in 2020	
	Min %/y	Max %/y	Min %/y	Max %/y
Overall	**0.500%**	**1.000%**	**0.500%**	**1.000%**
Inappropriate grazing, overgrazing and under-grazing	0.500%	1.000%	0.500%	1.000%

b) Degradation levels from the cumulative impacts of each key pressure

Degradation source / key pressures	Baseline 2010 levels		Maximum level for 15% restoration		Expected in 2020		% restoration required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Overall	**95.0%**	**100.0%**	**80.8%**	**85.0%**	**95.0%**	**100.0%**	**14.3%**	**15.0%**
Separation from the sea and impoundment	50.000%	80.000%	42.500%	68.00%	50.000%	80.000%	7.500%	12.000%
Inappropriate grazing, overgrazing and under-grazing	50.000%	80.000%	42.500%	68.00%	50.000%	80.000%	7.500%	12.000%

c) Conversion rates

Conversion causes	Conversion per year 2000-2010		Conversion per year 2010-2020		Total converted over 2000-2020		% re-creation required in 2020	
	Min %	Max %	Min %	Max %	Min %	Max %	Min %	Max %
Afforestation	0.010%	0.100%	0.010%	0.100%	0.200%	2.000%	0.030%	0.300%

6.11.6 Key management and restoration measures

Restoration of coastal habitats: general principles and generic costs

The restoration of Europe's coastal ecosystem requires **large-scale interventions** for restoring or creating coastal habitats. Restoration may require managed realignment, removal of hard infrastructure such as concrete sea defences or military installations, large-scale sediment supply and re-profiling, and new breakwater or reef construction. Most coastal restoration and habitat re-creation projects involve complex changes in hydrology, morphology, and geochemistry, at a range of spatial scales, and are subject to significant uncertainties. For example, scientific understanding of the links between sub-surface sediment structure, hydrology and biogeochemistry in restored saltmarshes is still poor (Spencer and Harvey, 2012). Changes in beach profiles (because of changes in the grain-size distribution of the nourishing sediments) can lead to changes in the hydrodynamics of the intertidal zone, leading to a reduction in diversity and abundance of the fauna and maybe even a permanent shift in the ecological community structure. Coastal habitats may also be significantly affected by changes in coastal areas quite a distance away, for example near-shore deep sediment dredging to improve port access may change tidal patterns and sediment flows to beaches and dunes outside the estuary system. The many economic and environmental interests involved, and the physical and ecological relations with other upstream and downstream systems underline the need for Integrated Coastal Zone Management (ICMZ) (Pickaver & Ferreira, 2008) and Integrated River Basin Management (IRBM).

Because of the uncertainties associated with ecosystem responses to management efforts, an **adaptive management** approach is needed, including continued monitoring of system responses to management efforts and research efforts to improve the understanding of ecosystem functioning (Spencer & Harvey, 2012). Drawing upon the experience of 51 restoration projects, Borja (Borja et al, 2010) conclude that full recovery of coastal marine and estuarine ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition and diversity may lag far beyond that period. Indeed, Elliot et al. (Elliott et al, 2007) emphasise that although recovery techniques are worthwhile if they can be carried out, they rarely (if ever) fully replace lost habitat.

Coastal habitat management for biodiversity conservation involves maintaining or restoring a **dynamic mosaic and/or zonation of habitat types,** driven by the continual **(re-)creation of open pioneer areas**. Dune systems are particularly short of pioneer habitat on beach lines, shifting dunes and dune slacks; salt marsh is particularly short of successional habitat in the higher zones but also requires sufficient intertidal flat to prevent loss to erosion; and coastal lagoons must be managed together with the surrounding habitats, including reed beds, sedges and wet meadows (without grazing and cutting, successional processes will lead to the lagoon growing over with rushes, reeds and other tall vegetation).

The management and restoration of coastal and marine habitats typically require measures in the field of regulation and spatial planning to **remove stressors and reduce pollution**, as well as habitat restoration or creation (Elliott et al, 2007). Stressor reduction includes measures to reduce eutrophication and chemical pollution, prevent habitat loss through development planning, restrict the introduction and spread of invasive species, and provide regulatory control of fisheries, recreation, shipping, and other activities that can damage habitats. The costs of these generic or regulatory measures are explored in Chapter 4.

Key maintenance and restoration measures for all coastal habitats
Management of human disturbance in coastal habitats is of importance to nesting birds, rare plants and invertebrates, especially on shingle beaches, dunes and coastal lagoon shorelines (Ausden, 2007). **Visitor management measures** to control disturbance and trampling include management of access points, creation and maintenance of paths, boardwalks and fencing, and information measures (Ausden, 2007). Legal measures for restricting access to vehicles, stakeholder consultation, and wider awareness campaigns are also important. The costs of these types of measures are not covered in this section.

Large-scale restoration of coastal habitats often requires the **removal of hard coastal defence infrastructure** such as sea walls, and other artificial infrastructure such as roads, car parks, military installations, or underground piping. The costs of these types of measures are not covered in this section.

Key maintenance and restoration measures for estuaries and coastal lagoons
Key maintenance measures for coastal lagoons include:

- Managing water levels and salinity levels so that they are optimal for desired species including birds and invertebrates, through **regulated tidal exchange, controlling freshwater inflows**, and/or **blocking outlets**. This requires **maintenance of weirs, dams, sluices, bridges** etc.
- Reducing impacts of eutrophication and managing succession by **removing algal blooms** and **mowing marginal vegetation, eg mosaic harvesting of reed beds**.
- Visitor and general management also includes maintaining paths, removing dead animals, maintaining bank-side structures etc.

Key habitat creation and restoration measures for coastal lagoons include (Borja et al, 2010):

- **Channel widening or narrowing**, or the **installation of weirs or sluices** to restore suitable salinity levels and hydrological conditions;
- **Sediment dredging** to counter eutrophication and succession;
- **Re-excavation** of overgrown lagoons;
- **Reprofiling** to create a variety of water depths, **creating islands** for nesting birds etc.

Key restoration measures for estuaries are partly covered under the separate component habitats – intertidal flats, saltmarsh, and sand dunes and beaches.

Key maintenance and restoration measures for beaches and intertidal flats
Measures to expand or recreate beach and dune habitats often involve 'soft coastal engineering' methods. On coastlines where managed re-alignment is being used to create or restore coastal habitats, **removal of sea defences and infrastructure** may be needed. On eroding coastlines, techniques include **construction of breakwaters** to reverse erosion, **sand replenishment or supply** on dunes or beaches or on the foreshore (using the sand transport forces of the sea and wind to promote 'natural' dune growth) (Worrall, 2005). It is important to ensure that the introduced sediments have the same sediment structure as

the existing beach; otherwise the interaction of waves and tides with the beach sediments will profoundly change the intra-sedimental fauna and the beach profile. Beach restoration often also requires large-scale **beach reprofiling**. Creation of new shingle beaches is particularly difficult and expensive due to shortage of material, and needs to be complemented with **planting or sowing vegetation** (Gilbert & Anderson, 1998).

Key maintenance measures for dune systems

Vegetated dunes and dune grasslands need **grazing** and/or **mowing** to slow succession. Grazing will require installation of **fencing** and **water supply**. Sheep are used to maintain a short turf on grey dunes for invertebrates and low-growing dune plants. Open range extensive grazing of cattle is a valuable management measure for maintaining a diversity of vegetation and disturbed patches, but will often require **shepherding** to avoid conflicts with recreational use. Rabbit grazing is an important influence on dune vegetation (especially on dune slacks where sheep will not graze), but **rabbit reintroduction** is often difficult in places where populations have been lost to disease.

Open dune habitat can be maintained through small scale **ploughing and sod cutting/turf stripping** to provide small patches of bare sand on vegetated dunes (Houston, 2008b). **Cattle grazing** can provide controlled small-scale erosion. Managed **path repositioning** can move the trampling of visitors, horse riders etc. across dunes in order to create new open habitat and allow colonisation of the closed off route.

Dune systems contain some valuable scrub habitats which should be maintained, but in some areas **scrub removal** will be necessary to maintain open dune vegetation. Dunes also need **regular cutting and clearance of invasive species**. The impacts of nitrogen deposition may need to be countered by an increase in the management interventions listed above in order to maintain habitat quality, including scrub removal and removal of turf or top soil (scraping) (Houston, 2008b).

Dune slacks are especially prone to succession if they are no longer maintained through the influence of freshwater and/or the scouring of wind, and may need to be **grazed**, **mown**, and/or periodically stripped of their top soil (**scraping**) (Houston, 2008a). However, pioneer dune slacks are susceptible to trampling damage by cattle and horses and may need to be protected by **fencing**. The periodic **creation of shallow open pools** in dune systems and salt meadows is important for the Natterjack Toad *Bufo calamita*.

Key restoration measures for dune systems

Large-scale recreation of dune coastal dynamics through removal of sea defences includes: **breaching and/or removal of sea dykes and walls** (sometimes with new banks inland); **removal of dune stabilisation structures** such as fences and groynes; **dune re-profiling** using bulldozers or diggers; sometimes it is necessary to build up the beach by importing sediment (**beach nourishment**). Heavily eroded sand dunes can be stabilised by **planting marram grass** and **erecting sediment traps**. Removal of forestry plantations requires **tree cutting** and **stump removal**, and may require use of manual labour rather than machinery (unless large scale dune reprofiling is also carried out). **Rewetting dune slacks** requires large-scale measures to raise the water table, such as drainage channel blockage or re-alignment on surrounding land (Grootjans et al, 2002).

Key maintenance and restoration measures for salt marshes
Intact saltmarsh habitat with natural hydrological dynamics should not be grazed (Doody, 2008). However, most saltmarshes and meadows have traditionally been grazed in the summer, and **grazing** needs to be continued to maintain a suitable vegetation structure for plant species diversity and for birds and vertebrates (Ausden, 2007). Reducing grazing does not usually recover the original vegetation. Restored or recreated saltmarsh is likely to need adaptive grazing or mowing management to control weedy species and to try to create conditions for species to recolonize (Wolters et al, 2005). The introduction of grazing requires the **installation of fencing, bridges, and cattle grids**[115]. Salt marshes may also be mown or burnt, but this is usually not recommended (Ausden, 2007; Doody, 2008).

Loss of intertidal flats and saltmarsh through erosion can be prevented or reduced by re-charging the intertidal flat with sediment by **depositing sediment** in offshore or on the intertidal flat where it will be re-distributed towards the shore. Some salt marshes may need to be protected from wave action, either offshore or on-site, using **breakwaters or other coastal defences** (Ausden, 2007).

Saltmarsh restoration and re-creation measures
Salt marsh and its intertidal habitat can be expanded or recreated by
- **managed re-alignment** - removing coastal defences to allow sea into previously reclaimed land, or
- **regulated tidal exchange** - regulating the flow of tidal water through previously impervious sea defences to allow controlled sea water flooding of previously reclaimed land (Ausden, 2007).

On levelled, consolidated reclaimed land, measures to re-introduce topographical variation are necessary to enable the saltmarsh creek system to reform, and to prevent the whole area becoming inundated (Ausden, 2007). This may involve **importing large quantities of material** to raise the level sufficiently, as impoundment and drainage causes oxidation of the saltmarsh sediments, as well as preventing accretion, and the soil level can drop significantly. It is also important to **in-fill any existing linear drains** on the reclaimed land to prevent the seawater draining away too quickly to deposit sediment. It is also interesting to **create some areas of saline lagoon**, especially for birds.

The key measures that address the key pressures on Europe's coastal ecosystems are summarised in Table 6-48.

[115] Along the Baltic, fencing coastal pastures means erecting the fence as far out as possible into the intertidal/near shore zone, so that the coastline is grazed right up to the water, but animals do not escape through shallow seawater.

Table 6-48 Key maintenance and restoration measures for coastal ecosystems and their required percentage coverage to address key pressures

Key pressure	Integrated Coastal Zone Management	Habitat (re-)creation	Break up hard coastal defence / managed realignment /RTE	Add coastal protection measures (soft)	sod cutting, disturbance	Removal of scrub & trees	Alter water levels and salinity	Sediment removal, dredging, re-profiling	Sediment addition, beach nourishment	Appropriate grazing regime and/or mowing	fencing, zoning, management of access	Comment
Habitat loss (coast erosion, infrastructure or residential development, reclamation)	100	100		50					100			Requires regulatory measures to prevent further loss
Habitat change due to lack of natural coastal dynamics – because of coastal defences, impoundment, separation from sea etc.	100	50	100		50			20	20	100		
Habitat change due to altered hydrology and salinity	100		50				100	50				
Afforestation (dunes only)				20		100						
Eutrophication	100				20	20						Requires measures to reduce pollution sources
Inappropriate vegetation management	100									100		Rabbit populations play important role
disturbance and damage due to human activities	100										100	Requires combination of physical in-habitat measures, regulatory and communication measures

6.11.7 Costs of maintenance and restoration

It was only possible to calculate costs for saltmarsh and for beach and dunes (it is assumed that this is made up of 31% beach with at least 11% shingle, 15% embryonic and shifting dunes, 23% vegetated dunes, 6% dune slacks and other wet dune habitat, and 25% dune grassland and machair). Costs data were sourced from nine Member States, but rely heavily on information from the Netherlands and the UK.

Ongoing costs of maintenance of coastal habitats

Maintenance costs of beaches

Beach nourishment to tackle habitat loss through erosion may cost 0.9 to 18 €/m3 of sediment, based on data from North Sea countries (RIKZ and Roelse, 2002). An average of 30 to 40 million € are dedicated to beach and foreshore nourishment along the Dutch coast each year; and more than 60 million € have been spent over the past 10 years for groins and dune regeneration at Saintes-Marie de la Mer (Petite Camargue) in France (EUROSION, 2004b). A calculation for 28 beaches along the Gulf of Cadiz in Spain indicated a yearly average cost of **2260** €/ha/y per ha of dry beach (berm) area nourished, to maintain the beaches over the course of a decade, ranging from 746 to 4645 €/ha/y on different beaches and different sediment sources (Munoz-Perez et al, 2001).

Excluding beach nourishment, beaches, shingles and embryonic dunes are estimated to be relatively cheap to maintain at 10 to 55 €/ha/y. In the UK, maintenance of vegetated shingle is financed under agri-environment at 63 €/ha/y whilst maintenance of restored shingle was costed at 325 €/ha/y in the Netherlands (GHK, 2006; Verheijen et al, 2009b).

For the purposes of this study, it is assumed that beach nourishment is carried out every other year to maintain the 20% of beach that is eroding, needing an average combined cost of 2000 €/ha/yr. It is assumed that the rest of the beach area requires 10 €/ha/yr in maintenance costs. This gives a combined maintenance cost of 408€/ha/yr of beach, and ***134 €/ha/yr*** *for maintaining beaches as a proportion of the total beach and dune area.*

Maintenance costs of dune systems

Dune landscapes with open dunes and grasslands are more expensive to maintain than beaches. Because of the high recreation importance of coastal habitats, costs of maintenance of dune systems are difficult to separate from costs of recreation management. A study compared combined nature conservation and recreation management of six different dune systems in the Netherlands and UK, and found a lower cost of 390 €/ha/y for a dune system managed by the state forestry organisation, and a high end cost of 960 €/ha/y for a dune system managed by a water supply company. This high cost is explained by the fact that the dune system is situated in the middle of a very densely populated area, resulting in higher recreation management costs, plus the production of drinking water in the dunes requires more intensive management to reduce the negative effects on its nature conservation value (Meulen et al, 2008). It is notable, however, that a UK dune system costs 1800 €/ha/y to manage (Meulen et al, 2008). Combined costs for ten different dune systems (the above plus others in NL & UK) averaged **377** €/ha/y (GHK, 2006; Meulen et al, 2008; Verheijen et al, 2009b), with an additional 30 €/ha for maintenance under nitrogen pressure (Oltmer et al, 2009). The English (UK) agri-environment payment for dune maintenance is currently **175** €/ha/yr (Natural England, 2009).

Some average costs for specific key maintenance measures are as follows:

- **Grazing of vegetated dunes** may cost between 56 and 106 €/ha/y, with an average cost of **77 €/ha/y** in three countries (FR, NL, UK) (Houston, 2008a; Houston, 2008b; UK Biodiversity Group, 2000; Verheijen et al, 2009b)
- Maintaining **fencing** on grazed dunes is costed at **75 €/ha/y** in the Netherlands (Verheijen et al, 2009b), (while reintroducing grazing on restored habitat may have a one-off cost of 500€/ha (UK Biodiversity Group, 2000))
- Removal of **unwanted (invasive) species** from dunes (*Senecio* and *Carduus* species) costs **6.3** €/ha/y in the Netherlands (Verheijen et al, 2009b)
- **Scraping open dunes** to maintain open habitat is costed at **32** €/ha/y in the Netherlands (Verheijen et al, 2009b)
- Maintenance of open dune habitat by **tree felling and scrub removal** has a cost of average 44 €/ha/y (NL data) (Verheijen et al, 2009b)
- **Mowing of grey dunes**, an infrequent management measure, costs 600 to 750 €/ha/y in France (Houston, 2008b)
- **Management of cropped machair** in the UK is funded under agri-environment at 275-338 €/ha/yr (Scottish RDP)
- **Fencing** as a measure to tackle disturbance by recreation may cost up to 5-48 €/m, leading to one–off costs of 1000 to 10,000 €/ha (for an average of 200 m/ha), while the maintenance cost may be 75 €/ha (Verheijen et al, 2009b)

For the purposes of this study, it is assumed that maintenance costs on dunes (including shifting dunes, vegetated dunes, dune grassland and machair) are 300 €/ha/yr, and ***189 €/ha/yr*** *within the total beach and dune area.*

Maintenance of **humid dune slacks** can be relatively costly. In the Netherlands, agreed costs are 1110 €/ha/ year (Verheijen et al, 2009b).

Some average costs for specific key maintenance measures are as follows:

- **Mowing dune slacks** costs 600 to 750 €/ha/y in France (Houston, 2008b) and 962 €/ha/y in the Netherlands, whilst extra mowing to control **invasive species** on dune slacks costs 6.3 €/ha/y (Verheijen et al, 2009b)
- Periodic maintenance **scrub removal on dune slacks** is costed at 1500-2200 in France (Houston, 2008a), which might need to be carried out twice over a decade on a third of the area, averaging at 100-146 €/ha/y for maintenance management
- **Scraping or sod cutting dune slacks** may cost up to 4700 €/ha in the Netherlands (Verheijen et al, 2009b), but is generally only applied on 25% of the area every 40 years, averaging out at 18-32 €/ha/y for maintenance management

Taking into account the costs for dune slack scrub clearance, it is assumed that combined maintenance costs on dune slacks are ***400*** *€/ha/yr, which is 24 €/ha/yr within the whole beach and dune area.*

*This gives a combined maintenance cost of **347€/ha/yr** for maintaining the total beach and dune area.*

Maintenance costs of salt marsh
Combined salt marsh management costs range between 0 and 130€/ha/y in the Netherlands and the UK (Natural England, 2009; Verheijen et al, 2009b).

Specific maintenance measures for saltmarsh include:

- **Grazing** of saltmarsh costs 38 €/ha/y under agri-environment in England (UK) with 88 €/ha/yr supplement for extensive grazing (Natural England, 2009), and 85 €/ha/y in the Netherlands (Verheijen et al, 2009b), but data from agri-environmental payments in Sweden (120-270 €/ha/y) suggest higher costs for grazing (Doody, 2008; Houston, 2008b). Better management of coastal saltmarsh by introducing a proper grazing management may cost up to 270 €/ha.
- The maintenance of **fencing**, bridges and cattle grids – the **exclusion of livestock** from saltmarsh is funded at 50 €/ha/y under agri-environment in England (UK) (Natural England, 2009)
- **Mowing** of saltmarsh may cost 20 €/ha/y on average in the Netherlands (Verheijen et al, 2009b), but agri-environmental payments in Sweden are 120-380 €/ha/y (Doody, 2008).
- Maintenance of coastal erosion protection measures such as **breakwaters or sea walls** are assumed to be part of baseline costs

*This study assumes an average maintenance cost of salt marsh of **130** €/ha/yr.*

Restoration and re-creation of coastal habitats
Addressing the degradation of coastal ecosystems will require **large-scale restoration and habitat creation** through managed realignment of coastal defences and through material supply. Realignment projects typically cost several millions, depending on location and size - typical examples cost 3 to 16 million € (Doody, 2008; EUROSION, 2004b). It was not possible to cost the removal of hard coastal infrastructure, such as sea walls, roads, piping or other concrete or metal structures.

For the UK it is estimated that the combined cost for **re-creating intertidal habitat** is 137,500 €/ha, part of an up to 37 million €/yr total cost for restoring all habitat lost by erosion in the UK (DEFRA, 2006). The cost of replacing the same area of saltmarsh that is being lost in the UK through **saltmarsh re-creation** is estimated to be around £33 million (41.2 million €) per year (assuming that double the lost area needs to be recreated, because 50% of the re-created area will be mud flat)[116] (DEFRA, 2006). Agri-environment payments in England (UK) pay 188 €/ha/y for **intertidal and saline habitat creation** through non-intervention where an unmanaged breach in the sea defences has already occurred, and up to 500 or 875 €/ha/y for intertidal and saline habitat creation on grassland or arable land respectively, through sea defence breaching, creak excavation and suitable management

[116] The capital costs data used in the UK national costs evaluation (DEFRA, 2006) are as follows; land purchase costs £9,675 - £15,000 per ha; pre-construction costs £130-150,000 per site (topographic survey, water level data, niche modelling, other design costs, EIA); construction of earthbank (including breach and mobilisation/demobilisation): £3,000 per linear metre.

(Natural England, 2009). A study quoted a cost for **shingle beach expansion** in the UK at 12,500 €/ha (GHK, 2006). Costs for **re-creation of coastal lagoons** through managed re-alignment in the UK range from 5,250 to 71,250 €/ha (DEFRA, 2002).

Costs of specific key measures for re-creation of intertidal habitat include:

- Based on UK data, the estimates are that **dike (re)construction** may cost 334,000 to 10 million € per km (including capital costs) (Turner et al, 2007).
- Re-creating intertidal habitat through the **positioning of wave breaks** may cost 250-1225 €/m of wave break (Scottish Natural Heritage, 2000), or up to 20,000 € per construction.
- **Land purchase** or agricultural opportunity cost are quoted between 5,600 and 16,800 €/ha (DEFRA, 2006; Lee, 2001; Turner et al, 2007).
- **Deconstruction of seawalls** or **installation of sluices**
- **Import of sediment** to build up sunken former agricultural land
- **Infilling of existing drains** to raise the water level on coastal wetlands (saltmarsh and dune slacks)
- Topographical reprofiling, including the creation of coastal lagoons
- Channel widening or narrowing, installation of sluices or dams on coastal lagoons
- Dredging of coastal lagoons

It is not possible from this data to estimate an overall cost for the restoration of coastal ecosystems as a whole in the EU-27. However, if the figure of 130,000 €/ha is applied to the whole area of coastal habitat estimated to be lost to erosion and development in the EU-27 between 2010 and 2020, the recreation of coastal habitat would cost **49 billion €**.

*For the purpose of this study, it is assumed that the additional costs of re-creation of saltmarsh are **1000 €/ha.***

Restoration costs of dunes and beaches

Combined restoration costs of existing dunes range from 275 to 4500 €/ha
Costs of restoration for dune systems vary widely, depending on the specific habitat type and starting point for restoration from arable land, forest or scrub overgrowth. **Restoration of sand dunes from forestry** was costed at 2500 €/ha (GHK, 2006) and 1000 €/ha in the UK (UK Biodiversity Group, 2000), and 2320 €/ha in Denmark (Houston, 2008b). The **removal of invasive** *Carpobrotus edulis* on 1.7 ha and **planting of native species** (with 600 plants/ha) cost 1923 €/ha on 12 ha of dunes with juniper scrub in Andalusia, Spain (LIFE03 NAT/E/000054). Restoration and protection on 3 km of coastal dunes with juniper scrub cost €79,736 for the **creation of artificial barriers** on 6.5 ha, and 6.5 km of **fencing and three boardwalks** in Sicily, Italy (LIFE02 NAT/IT/008533). Restoration of 55 ha of badly degraded dunes with juniper in Spain cost 21,378 €/ha, including removal of exotic plants, planting of native species, and **removal of infrastructures** (30,159 m^2 roads and car parks, 215 sanitation manholes, 85 drinking water boxes and 3,500 m of pipes) (LIFE04 NAT/ES/000044).

Some specific measures for dune restoration are:

- **Beach and dune reprofiling** costs 10 to 200 €/m in the UK (Scottish Natural Heritage, 2000).
- **Creation of coastal vegetated shingle and dune** on arable land or grassland in England (UK) is currently funded under agri-environment at 250 to 400 €/ha/yr respectively (Natural England, 2009).
- **Planting grasses for dune stabilisation** can cost from 2.5 to 25 €/m in the UK (Scottish Natural Heritage, 2000)
- **Felling of scattered trees and scrub removal** for open dune restoration has an estimated cost of 228 to 1250 €/ha, averaging 844 €/ha, based on data from UK, NL and DK (GHK, 2006; Houston, 2008b; UK Biodiversity Group, 2000)
- **Scrub removal from overgrown dune slacks** was costed at 3500-4500 €/ha in France (Houston, 2008a)
- **Restoration management** of sand dunes is financed at 175 €/ha/yr under agri-environment in England (UK) (Natural England, 2009)

*This study assumes that an average overall restoration cost of dunes is **1000 €/ha**.*

Additional important measures are often required to reduce **coastal ecosystem fragmentation**. This may require the restoration of habitat to increase the size of habitat patches and, where necessary to create habitat corridors and stepping stones to restore ecological connectivity amongst habitat patches. However, to avoid double counting of costs, such measures are not further considered or costed here as they are likely to be addressed to some extent through the restoration of 15% of degraded areas of the ecosystem, especially if strategically planned through green infrastructure support measures (see Chapter 3).

6.11.1 Estimated additional costs of achieving Target 2 for the ecosystem

The following section draws on the above review of ecosystem extent, degradation levels, key pressures and the cost of measures to estimate the total additional costs of maintaining the ecosystem (ie preventing further losses and degradation) and achieving the 15% restoration and re-creation targets. The estimated costs are additional to those for all measures that are expected to be taken up to 2020 (as described above under the reference scenario). However, it was only possible to calculate costs for the saltmarsh and the beach and dunes areas. The intertidal area requires some restoration measures (eg removal of infrastructure), but does not generally require measures that are separate from the maintenance and restoration measures of the aligning saltmarsh or beach or dune. As explained above, it was not possible to divide up the estuary area into its component saltmarsh, beach or dune and open water components to enable the application of relevant measures and costs. Costs data for coastal lagoons were sparse, and often classified under lake restoration (see previous section).

Two estimates are provided for beach and dune ecosystem maintenance and restoration costs based on two methods: firstly the overall level of ecosystem degradation and average combined costs of maintenance and restoration, and secondly, the proportion of the ecosystem that is impacted by key pressures and the costs of key measures that address each of the key pressures. Due to inadequate data, only one estimate is provided for beach and sand dune re-creation costs and for all saltmarsh costs. These estimates are based on overall degradation levels and combined measures. See section 3.4 for a detailed account of the methodology and its assumptions.

The expected costs of maintaining the beach and dune ecosystem and restoring 15% of degraded areas of beach and dune ecosystem under the 2020 reference scenario is shown in Table 6-49 and Table 6-50 and for saltmarsh in Table 6-51. The results indicate that the estimated additional annual cost of **maintaining** the ecological condition of beaches and sand dunes is between €563,000 and €2,250,000 according to overall degradation rates and combined measures costs, or between €352,000 and €1,400,000 based on key pressures and the specific costs of key measures.

The estimated additional average annual cost between 2010 and 2020 of achieving the **restoration target** is between €4.5 and 7.1 million, according to overall degradation rates and combined measures costs, or between €5.9 and 8.0 million based on key pressures and the specific costs of key measures.

The estimated additional average annual cost between 2010 to 2020 of achieving the **re-creation target** for beaches and sand dunes is between €9,740 and €97,00, according to overall degradation rates and combined measures costs.

It is important to note that these stated costs are low because it is assumed that improved regulation (eg through ICZM processes) could play a major part in achieving Target 2 for coastal ecosystems. Furthermore, Member States already have substantial on-going measures to address coastal erosion. These include 'soft' engineering of coastline protection, for example through beach nourishment by cyclical sediment transfer, as well as the maintenance of 'hard' sea defences. The EUROSION project estimated that public expenditure in the EU dedicated to coastline protection against the risk of erosion and flooding reached an estimated €3,200 million in 2001 (EUROSION, 2004a). This expenditure mainly reflects activities to protect coastal assets at imminent risk from coastal erosion, and does not reflect the hidden costs induced by flooding, saline intrusion, etc. The same authors quote another study that estimated the total cost of coastal erosion[117] for EU coastal states between 1990 and 2020 would add up to over €161,000 million, corresponding to an average of 5,400 million €/yr (EUROSION, 2004a).

It can therefore be assumed that much more restoration and re-creation could be achieved, with no additional costs, if on-going spending on coastal protection measures is directed to more flexible coastline management measures.

[117] The estimated direct coastal protection cost for European coastal states between 1990 and 2020 to exceed €120,000 million, with an additional (indirect) cost of over €41,000 million. Figures derived from the Global Vulnerability Assessment. WL Delft Hydraulics / Rijkswaterstaat, 1993.

Table 6-49 Projected additional costs (€) in 2020 of achieving Target 2 in beach and dune ecosystems on the basis of the current costs of combined measures and expected overall degradation extent in 2020

Important notes: All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. See section 3.5 for further explanation. Ecosystem area = 324,693 ha

	Cost	Area required over		Cost in 2020*	
Activity	€/ha	Min	Max	Min	Max
Maintenance required in 2020*2	347	0.50%	2.00%	563,000	2,250,000
Restoration required 2010-2020	1,000	14.00%	22.00%	4,550,000	7,140,000
Re-creation required 2010-2020	1,000	0.03%	0.30%	9,740	97,400

Table 6-50 Projected additional costs (€) in 2020 of achieving Target 2 within beach and dune ecosystem on the basis of current costs of key maintenance and restoration measures and expected key pressure levels in 2020

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Duplicate costs are removed where "Included below" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 324,693 ha

ANNUAL ONGOING MAINTENANCE NEEDS IN 2020

	% Area at risk in 2020						Annual costs in 2020	
Key Pressure	**Min %/y**	**Max %/y**	**Key measure**	**Current annual costs (€/ha/y)**	**% area applied to**	**Instrument**	**Min**	**Max**
ANNUAL ONGOING MAINTENANCE NEEDS IN 2020								
Lack of natural coastal dynamics	?	?	ICZM	0	100%	regulation	-	-
Lack of appropriate vegetation management	0.50%	2.00%	mowing dune slacks	600	6%	AEM	58,400	234,000
Lack of appropriate vegetation management	0.50%	2.00%	sod cutting	30	2%	AEM	974	3,900
Lack of appropriate vegetation management	0.50%	2.00%	grazing & fencing etc	150	100%	AEM	244,000	974,000
Lack of appropriate vegetation management	0.50%	2.00%	scrub control	100	30%	AEM	48,700	195,000
MAINTENANCE TOTAL							***352,000***	***1,410,000***

ADDITIONAL RESTORATION NEEDS TO REVERSE CURRENT DEGRADATION

Key Pressure	% Area degraded in 2020		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Lack of natural coastal dynamics	12.5%	16.3%	restoration grazing / mowing	175	63%	AEM	447,000	582,000
Lack of natural coastal dynamics	12.5%	16.3%	beach expansion & nourishment	10000	10%	Other	4,060,000	5,280,000
Lack of natural coastal dynamics	12.5%	16.3%	scrub removal on open dunes	800	25%	AEM	812,000	1,060,000
Lack of natural coastal dynamics	12.5%	16.3%	beach & dune reprofiling	1000	10%	Other	406,000	528,000
Lack of appropriate vegetation management	1.50%	6.00%	scrub removal from dune slacks	3000	10%	AEM	146,000	584,000
RESTORATION TOTAL							***5,870,000***	***8,030,000***

ADDITIONAL RE-CREATION NEEDS TO REVERSE CONVERSION SINCE 2000

Key Pressure	% Area converted per year		Key measure	Current one-off costs (€/ha)	% area applied to	Instrument	Average annual costs over 2010-2020	
	Min	Max					Min	Max
Habitat loss, afforestation, erosion and biult developments	0.03%	0.30%	beach & dune re-creation	1000	100%	LIFE, other	9,740	97,400
							-	-
RE-CREATION TOTAL							***9,740***	***97,400***

Table 6-51 Projected additional costs (€) in 2020 of achieving Target 2 for saltmarshes based on overall baseline and expected 2020 degradation extent and combined measures

All costs are gross costs in Euros and based on current costs estimated from as close to 2012 as possible. Costs are all additional to existing measures, and expected measures (ie the reference scenario) up to 2020. Costs and totals are rounded to three significant figures. Costs and totals are rounded to three significant figures. Costs do NOT include supporting actions (see chapter 4), wide-scale / at source measures to address pollution impacts (see chapter 5), or species-specific costs etc (see section 3.4.4). The area requiring maintenance and restoration is adjusted to take account of expected conversion from 2010-2020. The 2020 annual maintenance area does not include maintenance requirements for re-created areas as in practice these areas will still be in early stages of development. Duplicate costs are removed where "Included below" is indicated in the Instrument column. See section 3.5 for further explanation. Ecosystem area = 297,982 ha

Activity	Cost	Area required over		Cost in 2020	
	€/ha	Min	Max	Min	Max
Maintenance required in 2020	130	0.50%	1.00%	194,000	387,000
Restoration required 2010-2020	1,000	14.25%	15.00%	4,250,000	4,470,000
Re-creation required 2010-2020	1,000	0.06%	0.38%	17,900	112,000

6.12 MARINE HABITATS

6.12.1 Description and extent

The CORINE land cover data measures all national territorial waters seaward of the lowest tide limit (see annex 1 for details of the definition). The extent of marine habitat for each Member State is shown in Table 6-52.

Table 6-52 Area of marine ecosystem CLC (hectares)

Data relate to 2006 CLC data except where indicated as * where data are for 2000. The estuary CLC is also discussed in the coastal ecosystems section.

Land cover (ha)	Sea and Ocean CLC 523
Austria	
Belgium	4,411
Bulgaria	20,476
Cyprus	37,692
Czech Republic	
Denmark	309,398
Estonia	123,525
Finland	637,683
France	284,275
Germany	178,350
*Greece**	65,088
Hungary	
Ireland	217,077
Italy	378,668
Latvia	28,714
Lithuania	5,009
Luxembourg	
Malta	8,931
Netherlands	66,204
Poland	35,988
Portugal	61,315
Romania	16,181
Slovakia	
Slovenia	1,835
Spain	327,470
Sweden	698,430
United Kingdom	793,844
TOTAL	**4,300,564**

6.12.2 Key restoration measures

Marine habitat restoration needs and options are very different to those of terrestrial habitats, as reviewed above. This section therefore provides a more general overview of the available information on the costs of restoring marine habitats.

The marine restoration measures reviewed range from fleet reduction interventions, to fishing gear modifications to improve selectivity, to establishing marine protected areas, to environmentally friendly aquaculture methods. Estimating the costs of these measures is fraught with difficulty for numerous reasons elaborated on below. Hence there has been no EU wide, sea basin, or even Member State level attempt to estimate the costs of marine restoration measures. This section attempts to outline the studies and information available and the issues therewith.

In terms of estimating the costs of gear modifications, the economic evaluation studies performed have been few, and scattered widely across the EU. Kronbak et al. (2009) undertook a bio-economic evaluation of implementing trawl fishing gear with different levels of selectivity. The evaluation was applied to the case of the Danish trawl mixed fishery in Kattegat and Skagerrak, which experiences a high level of discards and by catches of several species. Four different kinds of selectivity scenarios were evaluated in comparison with a baseline. Table 6-53 presents the costs of obtaining the gear types for the different scenarios. It is clear that depending on the choice of gear, the cost of the measure varies widely. Similarly, Macher et al. (2008) conducted a cost benefit analysis of improving selective measures in a single species fishery in the case of the Norway lobster in the Bay of Biscay, and again, costs vary.

Table 6-53 Costs per trawl of obtaining selective gear (data obtained through interviews with the Danish Fishermen's Association)

Scenario 1	Scenario 2	Scenario 3	Scenario 4
90 mm codend with mesh panel	90 mm codend with grid	100 mm codend	120 mm codend
5000 DKK	13 500 DKK	12 500 DKK	12 500 DKK

Source: (Kronbak et al, 2009)

According to the FP6 project NECESSITY, which analysed modified fishing gear and practices to reduce bycatch in trawl fisheries, "a change in gear will entail higher investment costs in the short-run, in particular if a new regulation is implemented at short notice and without derogations. If the changes are introduced over a period of time making it possible to depreciate the original gear, extra costs will only occur if the new gear has a design that makes it permanently more expensive than the original. Certain types of gear, for example, with panels (windows with larger mesh sizes) may not be more expensive while gear with grids (steel or fibreglass devices) may be permanently more expensive" (Frost et al, 2007). The authors add that "makers of fishing trawls have indicated that the cost of gear changes intended to improve selectivity may be accomplished at costs less than €3,000 per trawl" (Frost et al, 2007).

Marine protected area costs are covered in relatively greater depth in the literature, however they are still not fully understood. Balmford et al. (2004) developed a model to predict total running (ie, maintenance) costs per unit area based on a survey of 83 MPAs worldwide. Using a similar survey, Gravestock et al. (2008) examined the income necessary for an MPA to achieve its management objectives, and Cullis-Suzuki and Pauly (2010) estimated the annual maintenance cost of the current global network of MPAs, and ranked the maritime countries of the world according to their financial investment in MPAs. In contrast, McCrea-Strub et al. (2011) investigated the attempts to quantify the cost of establishing MPAs in the first place, however their search for studies regarding establishment costs produced only one peer-reviewed study, which provided a detailed analysis of the establishment and maintenance costs for six MPAs in the Philippines. They suggest a number of reasons for the lack of literature on this topic:

- the duration of the establishment phase is difficult to define and can vary substantially between MPAs;
- funding for planning and development during this phase is typically derived from multiple sources located within and/or outside of the host country;
- the establishment period precedes the formation of a dedicated management entity usually responsible for the maintaining financial records; and
- records, if any, of one-off start-up costs are more likely to be lost over time.

Nevertheless, the authors attempted to undertake a cost estimate, the results of which are presented in Table 6-54, however of the thirteen MPAs used in the analysis only two are within the EU, and these are overseas territories of the Netherlands, so not actually in a European sea basin.

Table 6-54 Approximate total establishment cost and total establishment cost per unit area of 13 MPAs

Site	Total establishment cost	
	(2005 USD)	(2005 USD per km2)
Bibilik MPA	20,518	102,591
Talisay MPA	7,528	22,950
CHICOP	1,583,455	3,192,450
Villahermosa MS	8,179	11,802
Tambunan MPA	18,198	17,668
MISSTAMPA	16,040	10,025
Pilar MPA	8,212	4,578
Saba MP	557,237	64,050
Bonaire NMP	1,145,058	42,410
Nha Trang Bay MPA	2,370,832	14,818
Seaflower MPA	14,795,169	228
Mariana Trench MNM	10,000,000	41
PMNM	34,800,000	96

Source: (McCrea-Strub et al, 2011)

Finally, aside from gear modifications and MPAs, Brown and MacFadyen (2007) applied a cost benefit analysis to evaluate relative cost effectiveness of different management responses to ghost fishing, including gear retrieval programmes, demonstrating for this measure as well that there is large variation in costs depending on the specifics of the measure.

7 THE TOTAL COSTS OF ACHIEVING TARGET 2 ACCORDING TO POTENTIAL PRIORITISATION SCENARIOS

7.1 Introduction

This chapter firstly develops four potential scenarios for prioritising and disaggregating the overall 15% restoration component of Target 2 and then uses these scenarios as a basis for the estimation of achieving Target 2 across all ecosystem types. In particular, it considers the merits of selecting restoration targets (including re-creation components) that differ amongst ecosystems. This component of this study therefore aims to provide some preliminary information on the range of potential outcomes and costs of restoration under various scenarios, which will aid the Commission's future development of a strategic framework for the setting of habitat restoration priorities. The objective of the framework will be to help Member States achieve Target 2 by establishing a common understanding of what each Member State is expected to achieve in terms of restoration, and then providing a common agreed set of criteria for prioritising actions at national and sub-national levels. However, this current study is in itself not starting the development of the strategic framework, because this is being addressed in a separate study.

7.2 The potential advantages and disadvantages of options for setting ecosystem-specific restoration targets

There are a number of potential objectives that need to be considered in the establishment of the costing scenarios for this study. These are briefly outlined below.

7.2.1 Equitable sharing of effort amongst Member States

The simplest framework for achieving the target would be for each Member State to restore 15% of its degraded ecosystems. However, as discussed in section 3.2, **it is important to note that the target relates to 15% of the amount of ecosystems that are degraded** (ie NOT 15% of each ecosystem that is considered to be degraded). Thus, the allocation of a 15% target to each Member State would probably not result in a relatively even spread of costs and benefits, because the amount of restoration undertaken would also depend on the proportion of degraded ecosystem in each Member State. Instead, the target would result in more effort being undertaken by those Member States that have large areas of degraded ecosystems. This would to some extent result in an allocation of effort that is in accordance with the polluter pays principle. However, some ecosystem degradation could be the result of external pressures, such as transboundary air or water pollution, which would result in some costs not falling on the polluter. Furthermore some degradation could result from past actions, resulting in current generations bearing the restoration burden. As a result the allocation of a 15% target to each Members State could result in disproportionally high burdens for some, and may not be therefore acceptable in terms of equitability.

However, the target for restoration applies to the EU as a whole, and therefore there is scope for varying the allocation of the target amongst Member States, for instance to help with burden sharing and (as described below) to enable targeting of particularly important ecosystems. The adjustment of targets for burden sharing could, for example, take into account degradation from wide-scale pollution impacts, and the ability for the country to undertake restoration measures (eg taking into account restoration costs and GDP per capita). If targeting of important ecosystems is undertaken then burden sharing adjustments might need to be made to take into account the distribution of habitats.

Another option to take account of inequitable burdens would be to provide additional targeted funding, rather than to adjust targets. The potential for targeting funding towards restoration priorities is considered to some extent in the review of funding instruments in chapters 8 and 9, but it is beyond the scope of this study to investigate the potential for targeting to specific Member States.

7.2.2 Targeting efforts to ecosystems that will provide the greatest benefits

Ecosystems, and especially their specific component habitats, vary in their importance for biodiversity conservation (eg in terms of their threat status, international importance, number of threatened species) and in their provision of ecosystem services. Therefore targeting restoration and re-creation to ecosystems and habitats that would provide the greatest benefits could increase the value for money from restoration. In this respect it is important to note that CBD Target 15 for ecosystem restoration clearly gives a high level of importance to enhancing carbon sequestration and storage in ecosystems.

In terms of biodiversity conservation, value for money could be increased by targeting measures that also contribute to Target 1 of the Biodiversity Strategy (see Box 1.1). This would entail focussing on Habitats Directive Annex I habitats that have high proportions with an unfavourable conservation status and that would benefit from restoration, for example because they are rare and/or highly fragmented.

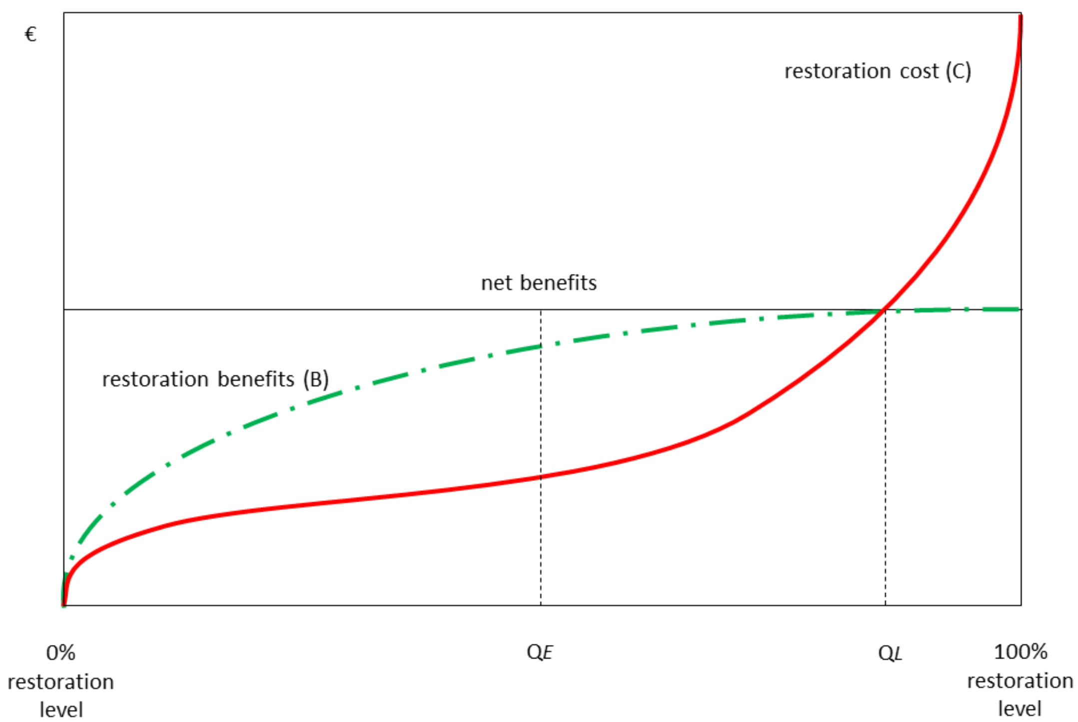
Restoration actions for nature conservation would undoubtedly contribute to the increased provision of many ecosystem services. However, a policy objective of focussing on increasing ecosystem service benefits would result in different restoration priorities. This could lead to more restoration of widespread and less natural ecosystems such as farmland and managed forests (eg to increase carbon storage and to protect soils and water resources). However, many service benefits (eg relating to water resources, pollination, soil erosion, protection from extreme events) are context and location specific. Thus what might be a priority in one country or local area in terms of ecosystem service provision might not be in another. Targeting of ecosystem restoration on the basis of generic ecosystem service values is therefore problematical (except for carbon, which has global-scale benefits) as a result of inadequate spatial information on ecosystem service values and beneficiaries. For such reasons the targeting of restoration according to ecosystem service is probably best tackled primarily at a national level.

7.2.3 Minimising costs / increasing cost-effectiveness

A policy objective may be to minimise the costs of achieving the 15% target across the EU. A simple EU strategy to keep costs low would be to focus on habitats that have the lowest generic restoration costs. However, restoration costs vary greatly amongst countries and depend on the condition of the ecosystem being restored and the degree of restoration undertaken. More importantly, minimising costs may not be the most cost-effective strategy as research summarised in section 2.8 shows that the economic benefits of ecosystem restoration can be greater than the costs, up to a certain point.

An important consideration is therefore how much should be spent on the restoration and re-creation of ecosystems that have been damaged or degraded? Whether it is 'good value for money' to have 15% or more restored as a standard target essentially depends upon the restoration cost and benefits curves. Figure 7-1 illustrates hypothetical examples of restoration cost and benefits curves; and shows how the cost-benefit ratio may vary according to the level of ecosystem restoration that is undertaken. In the figure the horizontal axis shows the amount of restoration necessary to restore the damaged ecosystem to a favourable condition, whilst the vertical axis shows the benefit of restored ecosystems in Euros, and also the Euro resource cost of restoring the ecosystems. Restoration costs (C) are an increasing function of the level of restoration, whilst restoration benefits (B) decline with the cumulative increase in restoration.

Figure 7-1 Cost-benefit considerations for ecosystem restoration and re-creation expenditure decisions



Cost and benefit curves for ecosystem restoration in the EU do not appear to have been published. However, it can be expected that in many situations the cost curves for ecosystem restoration will increase rapidly at first, but then the rate may decline as learning and economies of scale bring benefits (as discussed in section 3.4.4). Costs then start to rise more rapidly as suitable and low-cost restoration sites are used. The shape of benefit curves will vary according to the benefits in question, as some may continue to increase up to full restoration (eg as would be the case with carbon sequestration) whilst others may reach an early plateau (eg ecotourism in a remote area). According to this hypothetical scenario, the maximum achievable level of benefit is at full restoration. However, there is an argument for suggesting that restoration should not proceed beyond the level that costs exceed benefits (whether measured economically or in other terms). This indicates a restoration position at Q_L.

The most economically efficient level of restoration of ecosystems is where the net benefits [B-C] is positive and maximised, and this is at point Q_E. For restoration levels greater than Q_E, marginal costs are greater than marginal benefits. There may therefore be beneficial trade-offs between restoration of ecosystems with higher levels of benefits or rates of restoration levels. To achieve the overall 15% target most efficiently, it may therefore be appropriate to restore different ecosystems to different levels. These cost – benefit relationships are also likely to differ amongst Member States, and perhaps regions within them, and therefore decisions on the cost-effectiveness of restoration may well be best made by each Member State, taking into account potential EU-wide benefits.

Lastly, it should be noted that strategies that aim to keep costs low or maximise cost-effectiveness at an EU level could result in high burdens for some Member States if they have a high proportion of those habitats that have low restoration costs. Target adjustments or additional funding resources may therefore be necessary to ensure equitable sharing of the restoration burden.

7.2.4 Increasing the availability of funding

There are a variety of ways in which ecosystem restoration may be funded (see chapters 8 and 9), which include the use of public funds, such as through the CAP, or through market-based mechanisms, such as payments for ecosystems services (PES). There is therefore some merit in targeting restoration towards those ecosystems and Member States where there are the greatest opportunities for funding restoration. This would probably result in high levels of restoration being undertaken for agricultural ecosystems, due to the relatively large budget for agri-environment measures etc. There is less certainty over what could be achieved through PES schemes or other market-based measures. However, PES schemes have been set up by water companies to restore habitats in order to reduce soil erosion and thereby improve raw water quality. There is also potential for funding restoration of carbon-rich ecosystems through carbon offsetting or trading.

7.2.5 Ensuring the feasibility of targeting with available data

A final and essential consideration for setting national targets is whether or not scientific and economic data are adequate and sufficiently reliable to implement desirable strategic prioritisation frameworks. This study has found that data on ecosystem restoration costs are patchy with few data available for many countries. Furthermore, restoration costs vary greatly depending on the country and the ecosystem being restored, its condition and required level of restoration. Reasonable data are available on the importance and condition of different habitats from a nature conservation perspective. But information on the values of ecosystem services and therefore the economic benefits of restoration is very limited, and tends to show that benefits are highly context-specific. Therefore, although some strategies might appear to be optimal in terms of the objectives listed above, it may not be possible to implement them with currently available EU wide data.

Data constraints are, however, most marked at an EU level. Some Member States have good national data that could be used to prioritise ecosystem restoration in terms of, for example, ecosystem service benefits. Also the mapping and assessment of ecosystem services (MAES study), that is taking place in the context of Action 5 of the EU Biodiversity Strategy, is designed to help address such data constraints.

7.2.6 Restoration prioritisation considerations for Member States

The objectives described above relating to ecosystem targeting, cost-effectiveness and the availability of funds and data may also be taken into account in the prioritisation of restoration actions within Member States. In addition to these, it may also be appropriate to consider other factors at a national or sub-national level, including:

- Spatial / landscape-scale benefits that may arise from restoration, such as the potential for restoration to contribute to increasing the coherence of Natura 2000 networks, broader ecological networks and green infrastructure initiatives. Restoration could include increasing the size of small and isolated sites to increase the viability of populations within them, or creating new patches of habitat in the wider environment to increase connectivity between sites and the coherence of the Natura / ecological network (thereby helping to implement Article 10 of the Habitats Directive).

- Potential synergies with other sectoral and socio-economic objectives such as those relating to agriculture, transport, energy, tourism / recreation and other aspects of sustainable development.

- Possible linkages to restoration work in neighbouring countries.

Table 7-1 considers the advantages and disadvantages of a number of options for national targets for ecosystem restoration, according to the objectives described above.

Table 7-1 Potential options for prioritising the restoration and re-creation of ecosystems as a basis for setting national targets

Key: Potential for achieving the objective: +++ = high; ++ = moderate; + = low y; ? uncertain, more analysis required.

Option	Equitable sharing of effort	Targeting for biodiversity	Targeting for ecosystem services	Funding availability	Cost-effectiveness	Feasibility with available data
1. Even distribution across all MS	++?	++	++	++	++?	+++
2. Even distribution across each biogeographical region	++?	++	++	++	++?	+++
3. Even distribution across all ecosystems	?	+	+	++	+	+++
4. Focus on the most degraded ecosystems	?	++	++	?	++?	++
5. Focus on ecosystems with lowest restoration costs	?	+?	++?	++?	+	++
6. Focus on ecosystems with highest biodiversity value	?	+++	++	++	++	++
7. Focus on ecosystems with highest ecosystem service value	?	++	+++	+++	++	+

Notes. Option 1, ie 15% restoration in each MS, with no further disaggregation. Option 2, ie as 1 with 15% in each MS, but restoration carried out to also achieve 15% in each biogeographical region. Option 3, ie 15% of each degraded ecosystem restored (with fixed or variable MS targets). Option 4, ie allocation weighted according to extent of each ecosystem degraded (eg 15% of each degraded component restored). Option 5, ie lowest cost in terms of achieving the 15% target. Option 6, eg focussing on Habitats Directive Annex I habitats that have high proportions with an unfavourable conservation status and that would benefit from restoration, eg to increase the viability of individual SACs and SPAs and the coherence of the network habitat (thereby contributing to Target 1). Option 7, eg focussing on ecosystems that provide the highest levels of ecosystem service benefit (eg carbon sequestration, water storage and recreational values).

Clearly some of these objectives are mutually exclusive, but others could be combined, perhaps with weightings applied to favour certain priority objectives. Indeed, the analysis suggests that it would be appropriate for the strategic prioritisation framework to be based upon a combination of options to achieve most objectives, and in particular ensure cost-efficiency and a reasonable spread of costs and benefits (in terms of biodiversity and ecosystem services, and geographical spread).

7.2.7 The scenarios to be used in this study

On the basis of the above considerations this study has developed the following prioritisation scenarios for disaggregating the overall EU target amongst ecosystem types and Member States:

A. An even distribution of habitat restoration and re-creation across all Member States and ecosystems, ie restoration of 15% of each degraded ecosystem in each Member State (options 1 and 3 combined).

B. Restoration and re-creation of ecosystems prioritised to focus on those with the lowest restoration costs (options 1 and 7 combined).

C. Restoration and re-creation focused on the most important ecosystems for biodiversity conservation at an EU-scale (options 1 and 5 combined), in which the proportion of Habitats Directive Annex I habitats in each ecosystem type is used as proxy indicator of importance.

D. Restoration and re-creation focused on those that provide the greatest benefits across all major ecosystem services (options 1 and 6 combined). As discussed above, ideally this would be based on ecosystem service benefits as assessed at a Member State level, but data are not readily available to carry this out at present. This scenario is therefore based on a preliminary semi-quantitative assessment of EU-scale ecosystem service values (see below).

E. Restoration and re-creation focussed on ecosystems that provide the greatest combined benefit in terms restoring biodiversity and ecosystem services, this being a combination of scenarios 3, 4 and 6 above (ie, option 1 combined with 4, 5 and 6 in Table 7-1). This is simply based on an average of the scores allocated to each ecosystem in scenarios C and D.

For each scenario the percentage of each ecosystem type that would be restored and re-created is firstly calculated, and then equally distributed amongst each Member State. Thus, if a scenario results in a target of x% restoration for ecosystem A, then each Member State would restore x% of the degraded area of that ecosystem within their territory. However, it is important to note that consistent and comprehensive information on Member States specific ecosystem degradation levels are not available, and therefore this analysis is based on an EU-wide assessment of degradation levels for each ecosystem type. Clearly, ecosystem degradation levels will vary amongst Member States and therefore **further analysis would be necessary to establish the actual amounts of restoration required in each individual Member State, and their associated costs**.

For each scenario, the total costs of achieving Target 2 are calculated by combining all the estimated ecosystem-specific costs. The cost estimates are based on the costs of specific key measures to address key pressures (except for sclerophyllous scrub, where it was only possible to estimate costs according to general degradation levels and the costs of combined measures), because they are considered to provide the most robust and realistic estimate. Two cost estimates are provided for each scenario based on the minimum and maximum ecosystem-specific degradation levels as estimated for 2020 in accordance with the reference scenario. In addition a sensitivity analysis is conducted on each of the scenarios to test the influence of variation in restoration costs amongst Member States. Ideally it would also be valuable to examine variation in costs according to the degree of restoration that is undertaken, but unfortunately insufficient information is available on the degree of restoration that is likely to be achieved by the various restoration measures identified in this study.

The scenarios are also developed and costed separately for restoration and re-creation components because of their differing costs and because re-creation is not required or feasible for all ecosystems. Re-creation is not required for ecosystems that are expected to show a net decrease between 2000 (the baseline year for ecosystem extent) and 2020 (ie arable, permanent crops, improved grasslands and forests). For ecological reasons significant re-creation is also not possible or would be prohibitively expensive for mires and lakes and rivers.

The methods and data used to develop restoration and re-creation scenarios B – E are further described below. Scenario A is not described here because it is the scenario of 15% restoration for each ecosystem used in the costings for each ecosystem described in detail in chapter 6.

Scenario B: Restoration and re-creation of ecosystems prioritised to focus on those with the lowest restoration costs

The first step in developing this scenario is to calculate the relative per hectare costs of restoring each ecosystem, as shown in Table 7-2 below. In most cases, the costs are based on the average key measure restoration costs, as estimated for each ecosystem in chapter 6. These cost estimates are used where available because they are considered to be more robust than the combined measure costs. However, due to inadequate data, combined measure costs were estimated for sclerophyllous vegetation, marshes and saltmarsh. The relative cost of achieving the 15% target for each ecosystem was then calculated by dividing the total average cost by 15% of the total area of the ecosystem. This does not provide a true per hectare cost of restoring degraded areas, because this is not possible for costs based on key measures (as they address pressures with varying degradation levels). But the cost per hectare calculation method is appropriate for the ranking of the overall ecosystem restoration costs, as required in this study.

Table 7-2 Restoration costs for each ecosystem and the proportion restored to meet Target 2 according to the low cost scenario

15% Restoration target according to 2010 baseline = 30,100,00 ha. See text for explanation

Ecosystem	Average area (ha) degraded*1 in		Cost per ha	% to be restored to meet target	Area (ha) restored
	2010	2020			
Arable ecosystems including temporary grasslands	94,260,323	97,300,978	455	0.0%	-
Permanent crops	8,928,512	10,416,598	668	0.0%	-
Agriculturally improved permanent grasslands	26,312,451	26,312,451	292	0.0%	-
Natural and semi-natural grasslands	17,251,665	17,088,913	108	10.6%	1,810,000
Forest	37,985,032	42,543,235	58	50.0%	21,300,000
Heathland and tundra	2,237,295	3,132,214	3	50.0%	1,570,000
Sclerophyllous vegetation	2,510,221	3,048,126	486	0.0%	-
Mires (bogs and fens)	6,715,892	6,715,892	19	50.0%	3,360,000
Inland marshes	264,858	370,801	293	0.0%	-
Lakes and rivers	4,006,658	4,006,658	18	50.0%	2,000,000
Dunes & beaches	227,285	251,637	204	0.0%	-
Saltmarsh	290,532	290,532	100	50.0%	145,000
Total	**200,990,724**	**211,478,035**			**30,100,000**

Notes 1. Based on overall degradation rates as indicated in the ecosystem assessments in chapter 6. 2. Cost per hectare to achieve the target, based on key measure costs for all ecosystem types other than sclerophyllous vegetation, marshes and saltmarsh which are based on combined measure costs, divided by 15% of the entire ecosystem area (see text for further explanation).

Scenario B assigns the restoration target for each ecosystem by firstly selecting the ecosystem that has the lowest per hectare restoration costs. A maximum restoration target for the ecosystem is set at 50% of the degraded area. This percentage is used because it is assumed that the cost benefits curve would be similar to that depicted in Figure 7-1. With moderate levels of restoration the per hectare costs would be relatively low because the focus could be on sites that are easiest to restore and costs would decline for a while as restoration area increases due to the economies of scale (see section 7.2.3 for evidence and discussion). However, there will come a point at which increasing the percentage of the ecosystem restored would become more costly per hectare because sites that are less easy to restore would need to be included. Such sites might, for example need to include additional measures or be in locations that are remote or difficult to work in. It is not known at which percentage restoration level the unit costs would increase significantly (thereby potentially becoming more excessive than other less restored ecosystems), but it is assumed for this study that it could be around 50%.

Therefore, the progressive allocation of the restoration target for each ecosystem proceeds until the addition of 50% of the degraded area of the next lowest unit restoration cost ecosystem would exceed the area required to meet the overall 15% target across all ecosystems. The target area for the last added ecosystem (ie semi-natural grassland in the table above) is then capped such that the target area is that required to complete the overall 15% target area, which is 31,419,065 ha (based on a summation of the average overall projected degradation amounts in 2020).

As a result, as shown in **Error! Reference source not found.**, under this scenario the overall 15% restoration target would be achieved at minimum cost by restoring up to 50% of the degraded areas of permanent crops, agriculturally improved permanent grasslands, heathlands and lakes and rivers and 5.34% of arable crops. The projected costs of this scenario are shown in section 0.

A similar process was followed to develop Scenario B re-creation targets for each ecosystem type, the results of which are shown in Table 7-3.

Table 7-3 Re-creation costs for each ecosystem and the proportion re-created to meet Target 2 according to the low cost scenario

15% Re-creation target according to 2010 baseline = 358,000 ha*3. See text for explanation

Ecosystem	Average net area (ha) converted 2000-2020	Re-creation cost (€) per ha	Priority Group	% restored to meet target	Area (ha) restored
Arable ecosystems including temporary grasslands	0			0.0%	-
Permanent crops	0			0.0%	-
Agriculturally improved permanent grasslands	0			0.0%	-
Natural and semi-natural grasslands	1,409,428	3000	4	0.0%	-
Forest	0			0.0%	-
Heathland and tundra	40,719	406	1	50.0%	20,400
Sclerophyllous vegetation	896,508	2000	3	35.9%	322,000
Mires (bogs and fens)*	126,643	NA		0.0%	
Inland marshes	31,783	2000	3	36.7%	11,700
Lakes and rivers	0			0.0%	-
Dunes & beaches	3,572	1000	2	50.0%	1,790
Salt marshes	4,321	1000	2	50.0%	2,160
Total	**2,512,973**				**358,000**

Note. *1. Costs based on combined measures for each ecosystem type as indicated in chapter 6. *2 Mire re-creation is considered to prohibitively expensive or not feasible. *3 Excluding a requirement to re-create mires.

Scenario C: Restoration focused on the most important ecosystems for biodiversity conservation at an EU-scale

There are a number of ways of in which the relative importance of each ecosystem in terms of its contribution to nature conservation objectives could be assessed in theory. For example, a fairly thorough assessment might consider the number of characteristic species that each holds and the status of those species (ie the number that are locally, regionally and globally threatened, and the biogeographical importance of their EU populations). However, such data are not readily available and such an analysis is beyond the scope of this study. Furthermore, no ranking of the overall biodiversity importance of EU ecosystems is known of that could potentially be drawn on for this study.

This scenario has therefore used the proportion of the area of each ecosystem type that consists of Habitats Directive Annex I habitats (ie habitats of Community interest) as a proxy measure of biodiversity importance. Although this is fairly crude, and is also affected by data gaps (eg Annex I data for Romania and Bulgaria) it has the advantage of being closely aligned to EU biodiversity conservation objectives. In particular it is of relevance to Target 1 of the Biodiversity Strategy, and therefore would illustrate the potential contribution that a restoration strategy focussed on Target 1 could make.

There are also a number of ways in which the results of the assessment of nature conservation importance could be used to prioritise the restoration of each ecosystem type (ie to disaggregate the 15% target amounts the ecosystems). In this study we have used the following relatively simple approach because it could be easily taken up by each Member State (if required) and is consistent with the broad data that are currently available. Four priority classes are established according to the distribution of the scores. The percentage of each ecosystem that would need to be restored in each priority group to meet the overall EU target is then distributed in roughly equal divisions for the priority groups.

The same general approach and process is used for Scenarios D and E below. However, it should be noted that the priority groupings and proportion allocated to each group cannot be consistent amongst Scenarios C, D and E because the area of degraded ecosystem varies amongst the ecosystems, and therefore the percentage allocation needs to vary accordingly to deliver the same overall amount of restoration (ie 15% across all degraded areas).

It should also be noted that this allocation approach results in relatively moderate prioritisation with some restoration of all ecosystems. This will probably help to ensure a reasonably even spread of restoration geographically thereby also avoiding potentially high burdens falling on some Member States.

Table 7-4 below presents the proportion of the area of each ecosystem type that comprises a habitat of European conservation interest, and summarises the results of the prioritised target allocation amongst ecosystems for Scenario C. Under this scenario, highest priorities are given to restoration of heathland and tundra, mires, dunes and beaches, and saltmarsh ecosystems, with 50% of the degraded area of each restored. Only a low priority is given to the restoration of arable, permanent crops and improved grasslands, with restoration of these ecosystems limited to 7% of their degraded area. However, due to their extensive area, the actual area of arable and improved grassland that would be restored is much greater than the area restored for most other ecosystem types.

Using the same priority groupings,

Table 7-5 provides the targets under Scenario C for the re-creation of ecosystems.

Table 7-4 The proportion of the area of each ecosystem type that comprises a habitat of European conservation interest, and the resulting priority groups and % restoration allocations for Scenario C

15% Restoration target according to 2010 baseline = 30,100,00 ha. See text for explanation

Ecosystem	**Average area (ha) degraded in 2020**	**CLC area**	**N2K area***	**% N2K**	Priority **Group**	**% restored**	Area (ha) **restored**
Arable ecosystems inc. temporary grasslands	97,300,978	121,626,223	0	0%	5	6.5%	6,320,000
Permanent crops	10,416,598	14,880,853	0	0%	5	6.5%	677,000
Agriculturally improved grasslands	26,312,451	29,236,056	0	0%	5	6.5%	1,710,000
Natural and semi-natural grasslands	17,088,913	32,550,311	9,398,025	29%	3	30.0%	5,130,000
Forest	42,543,235	151,830,053	27,417,983	18%	4	20.0%	8,510,000
Heathland and tundra	3,132,214	8,949,182	9,888,150	110%	1	50.0%	1,570,000
Sclerophyllous vegetation	3,048,126	8,602,315	3,409,230	40%	3	30.0%	914,000
Mires (bogs and fens)	6,715,892	7,675,305	9,823,933	128%	1	50.0%	3,360,000
Inland marshes	370,801	1,059,432	437,753	41%	3	30.0%	111,000
Lakes and rivers	4,006,658	10,684,422	7,573,064	71%	2	40.0%	1,600,000
Dunes & beaches	251,637	324,693	344,764	106%	1	50.0%	126,000
Salt marshes	290,532	297,982	353,535	119%	1	50.0%	145,000
Total	**211,478,035**	**387,716,827**	**68,646,437**				**30,200,000**

Notes * From Natura 2000 database according to Member States, no data are available for Romania and Bulgaria. Some NK% proportions are >100%, due to inconsistencies in the underlying data, but this does not significantly affect the ranking.

Table 7-5 The proportion of the area of each ecosystem type that comprises a habitat of European conservation interest, and the resulting priority groups and % re-creation allocations for Scenario C

15% Re-creation target according to 2010 baseline = 358,000 ha[*3]. See text for explanation

Ecosystem	**Average cumulative**	**CLC area**	**N2K area***	**% N2K**	**Priority Group**	**% re-created**	**Area (ha) re-**

	area (ha) converted 2000-2020						created
Arable ecosystems including temporary grasslands	0	121,626,223	0	0%	5	0.0%	-
Permanent crops	0	14,880,853	0	0%	5	0.0%	-
Agriculturally improved permanent grasslands	0	29,236,056	0	0%	5	0.0%	-
Natural and semi-natural grasslands	1,409,428	32,550,311	9,398,025	29%	3	14.6%	206,000
Forest	0	151,830,053	27,417,983	18%	4	0.0%	-
Heathland and tundra	40,719	8,949,182	9,888,150	110%	1	35.0%	14,300
Sclerophyllous vegetation	896,508	8,602,315	3,409,230	40%	3	14.6%	131,000
Mires (bogs and fens)*1	126,643	7,675,305	9,823,933	128%	1	0.0%	
Inland marshes	31,783	1,059,432	437,753	41%	3	14.6%	4,640
Lakes and rivers	0	10,684,422	7,573,064	71%	2	0.0%	-
Dunes & beaches	3,572	324,693	344,764	106%	1	35.0%	1,250
Salt marshes	4,321	297,982	353,535	119%	1	35.0%	1,510
Total	**2,512,973**	**387,716,827**	**68,646,437**	**7**	**34**	**1**	**358,000**

Note. *1. Mire re-creation is considered to prohibitively expensive or not feasible. *2 Excluding a requirement to re-create mires.

Scenario D: Restoration focused on ecosystems that provide the greatest benefits across all major ecosystem services

Section 2.9 of this report, presented some case studies that assessed the effects of ecosystem restoration in terms of ecosystem service benefits. However, it is clear from the review that evidence of ecosystem service benefits is fragmentary because few restoration initiatives appear to undertake monitoring or publish the reposts of such impacts. It is therefore clearly not possible within this study to collate sufficient data on restoration benefits to develop cost-benefit curves (as in **Error! Reference source not found.**) that could e used to investigate restoration scenarios.

Despite the lack of direct data on restoration benefits, some general assessments of the ecosystem service provision have been carried out in recent years and these can be used to develop an indirect proxy estimate of the likely potential benefits of ecosystem restoration. It is anticipated that further work on mapping and assessment of ecosystem services (under Action 5 of the Biodiversity Strategy) by the MAES Working Group and a current TEEB Europe study will be able to provide data that can be used to refine the assessment in the future. In fact an initial discussion paper (Maes et al, 2012) outlines the intention of the

MAES Working Group to adopt the RUBICODE Matrix that cross-tabulates broad ecosystems and their services, their relative importance in delivering ecosystem services and trends in ecosystem services change. Furthermore, the working group will set out to assist the design of prioritisation criteria for restoration and determine at which scale benefits may be delivered in a cost-effective way.

For the purposes of the development of an ecosystem service prioritised scenario for this current study, we have drawn on work by Burkhard et al (2009), which provided a semi-quantified assessment of the different CLC types capacities' to provide ecosystem services. A modified version of this resulting matrix, depicting the relevant land cover types used in this study and 22 ecosystem services, is provided below in Table 7-6.

Table 7-6 Assessment of potential provision of ecosystem services in Europe according to land cover type

Capacities to provide each service: 0 = no relevant capacity; 1 = low relevant capacity; 2 = relevant capacity; 3 = medium relevant capacity; 4 = high relevant capacity; 5 = very high relevant capacity. **Ecosystem grouping** indicates the ecosystem typology as used in this current study: 1 = Agro-ecosystems that include crop land; 2 = Permanent crops; 3 = Agriculturally improved permanent grasslands; 4 = Natural and semi-natural grasslands; 5 = Forest and transitional woodland-scrub; 6= Moorland, tundra and heathland; 7 = Sclerophyllous vegetation; 8 = Mires, bogs and fens; 9 = Marshes; 10 = Rivers and lakes; 11 = Coastal wetlands – beaches, estuaries and lagoons; Inland and coastal dunes; 12 = Marine habitats.

Ecosystems		Provisioning services											Regulating services									Cultural services				
Ecosystem grouping	**Ecosystem type**	**Crops**	**Livestock**	**Fodder**	**Capture Fisheries**	**Acquaculture**	**Wild Foods**	**Timber**	**Wood Fuel**	**Energy (biomass)**	**Biochemicals / medicine**	**Freshwater**	**Local climate regulation**	**Global climate regulation**	**Flood protection**	**Groundwater recharge**	**Air quality regulation**	**Erosion regulation**	**Nutrient regulation**	**Water regulation**	**Pollination**	**Recreation & tourism**	**Count**	**Sum**	**Proportion of total potentiall score**	**Proportion of potential score by ecosystem group**
1	Agriculture & natural vegetation	3	3	2			3	3	3	3	1		3	2	1	2	1	3		1		2	16	36	0.34	0.25
1	Annual and permanent crops	5	5	5						5	1		2	1	1	1	1	1				1	12	29	0.28	
1	Non-irrigated arable land	5	5	5						5	1		2	1	1	1						1	10	27	0.26	
1	Permanently irrigated land*	5	5	2						5	1		3	1	1							1	9	24	0.23	
1	Complex cultivation patterns	4		3							2		2	1	1	1						2	8	16	0.15	
2	Fruit trees and berries	5						4	4				2	2	2	2	2	2	1	1	5	5	13	37	0.35	0.23
2	Olive groves	4						4	4				1	1		1	1	1	1	1		5	11	24	0.23	
2	Vineyards	4							1				1	1		1						5	6	13	0.12	

3	Pastures		5	5									1	1	1	1		4				3	8	21	0.20	
4	Natural grassland		3				2						2	3	1	1		5	5	5		3	10	30	0.29	0.25
5	Broad-leaved forest			1			5	5	5		5		5	4	3	2	5	5	5	5	5	5	15	65	0.62	0.45
5	Coniferous forest			1			5	5	5		5		5	4	3	2	5	5	5	5	5	5	15	65	0.62	
5	Mixed forest			1			5	5	5		5		5	4	3	2	5	5	5	5	5	5	15	65	0.62	
5	Agro-forestry areas	3	3	2				3	3				2	1	1	1	1	2	1	1	3	3	15	30	0.29	
5	Transitional woodland scrub		2				1		2				1								2	2	6	10	0.10	
6	Moors and heathland		2				1		2	5			4	3	2	2			3	4	2	5	12	35	0.33	0.33
7	Sclerophyllous vegetation		2				1		2		3		2	1	1	1					2	2	10	17	0.16	0.16
8	Peatbogs									5			4	5	3	3			3	4	2	4	9	33	0.31	0.31
9	Water courses				3		4					5	1	0	2	1			3	3		5	10	27	0.26	0.23
9	Water bodies				3		4					5	2	1	1	2			1			5	9	24	0.23	
10	Inland marshes		2	5									2	2	4	2			4				7	21	0.20	

Ecosystems		Provisioning services											Regulating services									Cultural services				
Ecosystem grouping	Ecosystem type	Crops	Livestock	Fodder	Capture Fisheries	Acquaculture	Wild Foods	Timber	Wood Fuel	Energy (biomass)	Biochemicals / medicine	Freshwater	Local climate regulation	Global climate regulation	Flood protection	Groundwater recharge	Air quality regulation	Erosion regulation	Nutrient regulation	Water regulation	Pollination	Recreation & tourism	Count	Sum	Proportion of total potential score	Proportion of potential score by ecosystem group
11	Estuaries				5	5	4			3					3				3	3		4	8	30	0.29	0.15
11	Coastal lagoons				4	5	4			3			1		4							5	7	26	0.25	
11	Salt marshes		2										1		5				2			3	5	13	0.12	
11	Beaches, dunes and sand plains									2					5	1						5	4	13	0.12	
11	Intertidal flats												1		5				1			4	4	11	0.10	
11	Salines												2									2	2	4	0.04	
12	Marine (seas and oceans)			1	5	5							3	5					5			4	7	28	0.27	0.27
	Green urban areas						1		1				2	2		2	1	2	1	1	1	3	11	17	0.16	0.16

* Ricefields were taken out from agro-ecosystems, as they are very small areas and likely to skew the overall results.

The assessments have been used to develop an average total score for each of the ecosystem types used in this study. Equal weighting is given to all the ecosystem services proposed by Burkhard et al (2009), therefore those which have both high provisioning and regulating (and to a lesser extent cultural) services score significantly higher. In addition, any overlaps or conflicts between enhancing the ecosystem service and biodiversity requirements is not considered.

Ideally it would be appropriate to also consider the degree to which the services provided by each ecosystem may recover following restoration. Therefore this issue was investigated as part of the literature review on ecosystem restoration benefits. Information on the proportion of carbon storage capacity of pristine habitats returned as a consequence of restoration was found for ‘mires, bogs and fens’, which suggested that restoration could lead to service recovery of 77%. However, no similar information appeared to be readily available for other ecosystems in comparable situations to those under investigation in this study. Therefore, in the absence of reliable and consistent data for each ecosystem the ranking of ecosystem service restoration benefits assumes that restoration would enable reasonably comparable levels of service recovery, at least in the long-term.

Table 7-7 presents the overall ranking for each ecosystem, and indicates the percentage of the ecosystem that will be restored according to the ranking. The allocations were made using a similar approach to that described above for scenario C. Under this scenario most restoration is focussed on forests, but also substantial restoration of natural and semi-natural grasslands, heathland and tundra and mires.

Using the same priority groupings, Table 7-8 provides the targets under Scenario D for the re-creation of ecosystems.

Table 7-7 The ranking of ecosystem services provided by each ecosystem type, and the resulting priority groups and % restoration allocations for Scenario D

15% Restoration target according to 2010 baseline = 30,100,00 ha. See text for explanation

Ecosystem	Average area (ha) degraded in 2020	Ecosystem service score	Priority Group	% restored	Area (ha) restored
Arable ecosystems including temporary grasslands	97,300,978	0.25	3	9.0%	8,760,000
Permanent crops	10,416,598	0.23	3	9.0%	937,000
Agriculturally improved permanent grasslands	26,312,451	0.20	3	9.0%	2,370,000
Natural and semi-natural grasslands	17,088,913	0.29	2	18.0%	3,080,000
Forest	42,543,235	0.45	1	30.0%	12,800,000
Heathland and tundra	3,132,214	0.33	2	18.0%	564,000
Sclerophyllous vegetation	3,048,126	0.16	4	3.0%	91,400
Mires (bogs and fens)	6,715,892	0.31	2	18.0%	1,210,000
Inland marshes	370,801	0.20	3	9.0%	33,400
Lakes and rivers	4,006,658	0.24	3	9.0%	361,000
Dunes & beaches	251,637	0.12	4	3.0%	7,550
Saltmarsh	290,532	0.12	4	3.0%	8,720
Total	**211,478,035**				**30,200,000**

Table 7-8 The ranking of ecosystem services provided by each ecosystem type, and the resulting priority groups and % re-creation allocations for Scenario D

15% Re-creation target according to 2010 baseline = 358,000 ha[*2]. See text for explanation

Ecosystem	Average cumulative area (ha) converted 2000-2020	Ecosystem service score	Priority Group	% restored	Area (ha) restored
Arable ecosystems including temporary grasslands	-	0.25	3	0.0%	-
Permanent crops	-	0.23	3	0.0%	-
Agriculturally improved permanent grasslands	-	0.20	3	0.0%	-
Natural and semi-natural grasslands	1,409,428	0.29	2	21.0%	296,000
Forest	-	0.45	1	0.0%	-
Heathland and tundra	40,719	0.33	2	21.0%	8,550
Sclerophyllous vegetation	896,508	0.16	4	5.0%	44,800
Mires (bogs and fens)[*1]	126,643	0.31	2	0.0%	
Inland marshes	31,783	0.20	3	12.0%	3,810
Lakes and rivers	-	0.24	3	0.0%	-
Dunes & beaches	3,572	0.12	4	5.0%	179
Salt marshes	4,321	0.12	4	5.0%	216
Total	**2,512,973**				**354,000**

Note. *1. Mire re-creation is considered to prohibitively expensive or not feasible. *2 Excluding a requirement to re-create mires.

Scenario E: Restoration focussed on ecosystems that provide the greatest combined benefit in terms of reversing contributing to biodiversity conservation and ecosystem services

The scenario is a combination of Scenarios C and D, and is summarised in Table 7-9 below, which presents the combined ranking for each ecosystem, and indicates the percentage of the ecosystem that will be restored according to the ranking. The allocations were made using a similar approach to that described above for Scenarios C and D.

In this scenario restoration is focussed on heathland, tundra and mires, but there is also a significant amount of restoration across all other ecosystems.

Using the same priority groupings, Table 7-10 provides the targets under Scenario E for the re-creation of ecosystems.

Table 7-9 The combined ranking of biodiversity priority and ecosystem services for each ecosystem type, and the resulting priority groups and % restoration allocations for Scenario E

15% Restoration target according to 2010 baseline = 30,100,00 ha. See **Error! Reference source not found.**

Ecosystem	Average area (ha) degraded in 2020	Priority Group			% restored	Area (ha) restored
		Natura	Ecosystem service	Average		
Arable ecosystems including temporary grasslands	97,300,978	5	3	4	10%	9,730,000
Permanent crops	10,416,598	5	3	4	10%	1,040,000
Agriculturally improved permanent grasslands	26,312,451	5	3	4	10%	2,630,000
Natural and semi-natural grasslands	17,088,913	3	2	3	20%	3,420,000
Forest	42,543,235	4	1	3	20%	8,510,000
Heathland and tundra	3,132,214	1	2	2	40%	1,250,000
Sclerophyllous vegetation	3,048,126	3	4	4	10%	305,000
Mires (bogs and fens)	6,715,892	1	2	2	40%	2,690,000
Inland marshes	370,801	3	3	3	20%	74,200
Lakes and rivers	4,006,658	2	3	3	20%	801,000
Dunes & beaches	251,637	1	4	3	20%	50,300
Saltmarsh	290,532	1	4	3	20%	58,100
Total	**211,478,035**					**30,100,000**

Table 7-10 The combined ranking of biodiversity priority and ecosystem services for each ecosystem type, and the resulting priority groups and % re-creation allocations for Scenario E

15% Re-creation target according to 2010 baseline = 358,000 ha[*2]. See text for explanation

Ecosystem	Average cumulative area (ha) converted 2000-2020	Priority Group			% restored	Area (ha) restore
		Natura	Ecosystem service	Average		
Arable ecosystems including temporary grasslands	-	5	3	4	0%	-
Permanent crops	-	5	3	4	0%	-
Agriculturally improved permanent grasslands	-	5	3	4	0%	-
Natural and semi-natural grasslands	1,409,428	3	2	3	18%	254,000
Forest	-	4	1	3	0%	-
Heathland and tundra	40,719	1	2	2	35%	14,300
Sclerophyllous vegetation	896,508	3	4	4	9%	80,700
Mires (bogs and fens)[*1]	126,643	1	2	2	0%	
Inland marshes	31,783	3	3	3	18%	5,720
Lakes and rivers	-	2	3	3	0%	-
Dunes & beaches	3,572	1	4	3	18%	643
Salt marshes	4,321	1	4	3	18%	778
Total	**2,512,973**					**356,000**

Notes. *1. Mire re-creation is considered to prohibitively expensive or not feasible. *2 Excluding a requirement to re-create mires.

7.3 The total costs of achieving Target 2 according each target disaggregation scenario

The costs of achieving Target 2, in addition to those associated with expected measures up to 2020 (ie the reference scenario), are presented below for each target disaggregation scenario, according to minimum and maximum estimated degradation levels. The estimates costs are the estimated expenditure required in 2020, which comprise the costs of annual maintenance measures and the one-off costs of restoration and re-creation measures assuming equal implementation of measures between 2010 and 2020 (ie restoration and re-creation costs are $1/10^{th}$ those of the total estimated requirement between 2010 and 2020). Costs are based on the current costs of measures and are not adjusted for anticipated inflation or other possible changes in the costs of measures in 2020.

The costs are firstly presented for maintenance measures only. These do not differ across the scenarios because Target 2 requires “the maintenance of ecosystems”, which is assumed to apply to the entire extent of all ecosystems and is therefore independent of the 15% restoration target and any disaggregation scenario. The estimated total costs of restoring and recreating each ecosystem are then presented separately, to show the implications of the differing scenarios and because they have been calculated using slightly different methods and assumptions (as described above).

By breaking down the total costs for each ecosystem according to the proportion of each ecosystem in each Member State (according to CLC data) it is possible to also provide estimates of the cost implications for each Member State according to each scenario. However, it should be noted that such estimates are based on overall EU degradation rates / pressures levels because comparable Member State level data are not available. Therefore costs would be lower than projected for Member States with lower than average levels of ecosystem degradation, and higher if degradation levels are above average.

Member State level costs are presented firstly on the basis of the average costs of maintenance, restoration and re-creation measures as calculated for each ecosystem in chapter 6. However, the actual cost of achieving Target 2 in each Member State will vary according to the purchasing power of their currency. For example, in some countries labour costs and the cost of land purchase may be much lower than the EU average and therefore allow management and restoration measures to be carried out at lower costs. To allow for this a second set of cost estimates are provided that are adjusted according to the differing purchasing power of the currency in each country.

The results provided in Tables 7-11 to 7-15 and Figure 7-2 ndicate that the estimated annual additional cost of **maintaining** the ecological condition of all ecosystems is between €618 and 1,660 million. The costs of maintaining each ecosystem vary considerably, largely due to their differing area. But, as discussed for each ecosystem type in chapter 6, costs also vary in relation to the need for management measures and the extent to which such requirements are already provided for through existing and expected measures. Hence the highest maintenance costs are for arable and forest ecosystems, requiring approximately €220 to 556 million and €196 to 488 million, respectively. Additional maintenance costs are also substantial for permanent crops, agriculturally improved grasslands, and natural and semi-natural grasslands. In contrast, there are no additional maintenance costs for lakes and rivers up to 2020 because it is anticipated that measures required under the WFD to maintain the condition of water bodies will be sufficient to achieve Target 2 maintenance needs.

Table 7-11: Total additional annual costs in 2020 of maintaining each ecosystem

Ecosystem type	Costs (€) according to degradation levels	
	Min	Max
Arable ecosystems including temporary grasslands	220,000,000	556,000,000
Permanent crops	52,400,000	175,000,000
Agriculturally improved permanent grasslands	77,600,000	191,000,000
Natural and semi-natural grasslands	43,200,000	178,000,000
Forest	196,000,000	488,000,000
Heathland and tundra	2,250,000	17,100,000
Sclerophyllous vegetation	14,300,000	35,900,000
Mires (bogs and fens)	8,140,000	10,400,000
Inland marshes	3,180,000	6,360,000
Lakes and rivers*	-	-
Dunes & beaches	352,000	1,410,000
Salt marshes	194,000	387,000
Total	**620,000,000**	**1,660,000,000**

Note. * It is anticipated that maintenance measures will be adequately provided through existing and anticipated measures required under the WFD.

Figure 7-2 Total additional annual costs in 2020 of maintaining each ecosystem

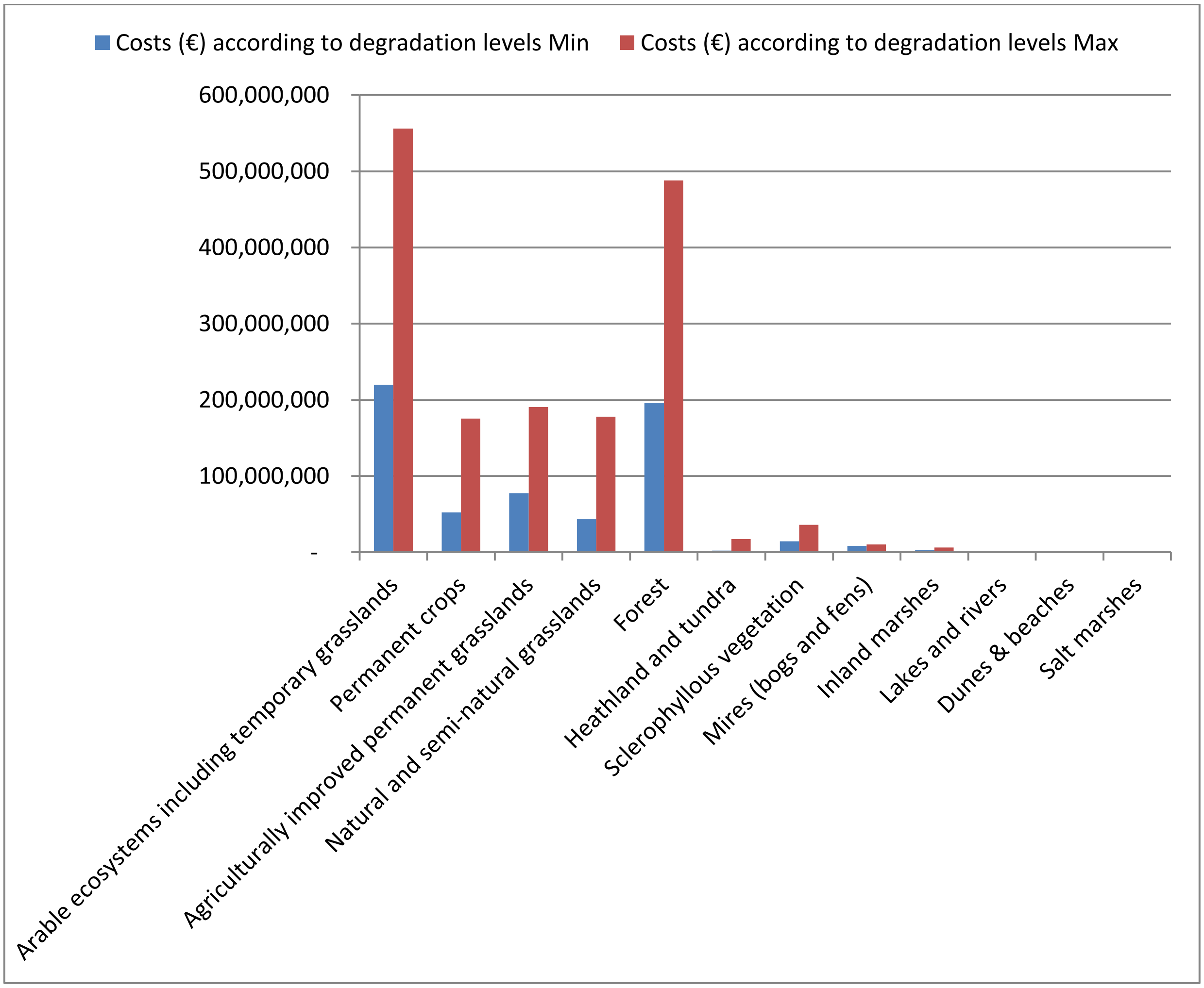


Table 7-12 Percentage of total maintenance costs comprised by each ecosystem

Colour shading reflects the proportion of the total cost attributed to each ecosystem type

Ecosystem type	Costs (€) according to degradation levels	
	Min	Max
Arable ecosystems including temporary grasslands	36%	34%
Permanent crops	8%	11%
Agriculturally improved permanent grasslands	13%	11%
Natural and semi-natural grasslands	7%	11%
Forest	32%	29%
Heathland and tundra	0%	1%
Sclerophyllous vegetation	2%	2%
Mires (bogs and fens)	1%	1%
Inland marshes	1%	0%
Lakes and rivers	0%	0%
Dunes & beaches	0%	0%
Salt marshes	0%	0%

The national costs of maintenance measures also varies considerably, principally in relation to the area of each Member State and the proportion of ecosystems within each that have costly maintenance requirements (Table 7-13).

The estimated national costs of maintenance measures are adjusted according to purchasing power in Table 7-14. Although this has little impact on the overall maintenance total (which is approximately 95% of the unadjusted total), it does significantly affect costs in some countries. Most notably costs are increased significantly for Denmark, Finland, Luxembourg and Sweden. In contrast costs are almost halved for Bulgaria, Hungary, Lithuania, Poland and Romania. According to these adjusted estimates the highest maintenance costs fall within France and Spain, where average (ie mid-point) costs are well over €100 million (Figure 7-3). Most other countries have mid-point costs in the 10’s of millions.

Table 7-13 Total additional annual maintenance costs in 2020 for each Member State

Member State	Costs (€) according to average EU degradation levels	
	Min	Max
Austria	10,400,000	29,700,000
Belgium	4,231,000	11,100,000
Bulgaria	16,200,000	44,400,000
Cyprus	1,430,000	3,930,000
Czech Republic	11,500,000	31,100,000
Denmark	6,800,000	17,400,000
Estonia	6,300,000	16,300,000
Finland	39,200,000	96,300,000
France	90,400,000	239,000,000
Germany	55,300,000	143,000,000
Greece*	?	?
Hungary	14,800,000	39,500,000
Ireland	13,500,000	33,300,000
Italy	47,200,000	133,000,000
Latvia	9,170,000	25,300,000
Lithuania	10,300,000	26,600,000
Luxemburg	413,000	1,060,000
Malta	54,100	152,000
Netherlands	5,870,000	15,100,000
Poland	49,200,000	128,000,000
Portugal	13,800,000	41,000,000
Romania	35,300,000	98,600,000
Slovakia	7,000,000	18,700,000
Slovenia	2,770,000	7,670,000
Spain	80,800,000	233,000,000
Sweden	49,000,000	124,000,000
United Kingdom	36,400,000	101,000,000
TOTAL	**617,000,000**	**1,660,000,000**

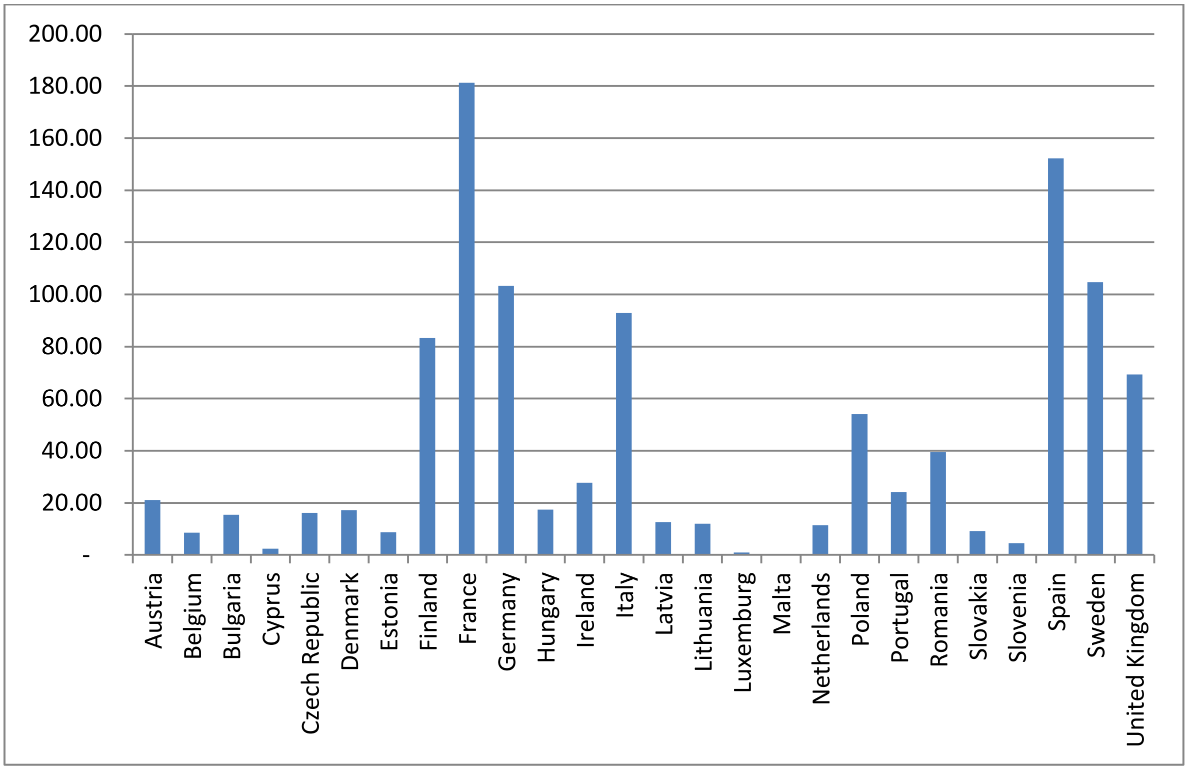
Notes. Cost estimates are based on general EU-wide degradation levels, and actual Member State costs would differ according to national ecosystem degradation levels. * Costs cannot be estimated for Greece due to inadequate CLC data.

Table 7-14 Total additional annual maintenance costs in 2020 for each Member State, with costs adjusted according to variation in Member State prices

Member State costs	Costs (€) according to average EU degradation levels		Price adjustment
	Min	Max	
Austria	10,900,000	31,200,000	1.05
Belgium	4,700,000	12,300,000	1.11
Bulgaria	8,240,000	22,600,000	0.51
Cyprus	1,280,000	3,520,000	0.90
Czech Republic	8,770,000	23,600,000	0.76
Denmark	9,590,000	24,600,000	1.41
Estonia	4,810,000	12,400,000	0.76
Finland	48,200,000	118,000,000	1.23
France	99,400,000	263,000,000	1.10
Germany	57,500,000	149,000,000	1.04
Greece	-	-	0.95
Hungary	9,460,000	25,300,000	0.64
Ireland	16,000,000	39,300,000	1.18
Italy	48,600,000	137,000,000	1.03
Latvia	6,700,000	18,400,000	0.73
Lithuania	6,720,000	17,300,000	0.65
Luxemburg	504,000	1,290,000	1.22
Malta	41,700	117,000	0.77
Netherlands	6,340,000	16,300,000	1.08
Poland	30,000,000	78,000,000	0.61
Portugal	12,200,000	36,100,000	0.88
Romania	20,800,000	58,200,000	0.59
Slovakia	4,970,000	13,200,000	0.71
Slovenia	2,390,000	6,600,000	0.86
Spain	78,300,000	226,000,000	0.97
Sweden	59,200,000	150,000,000	1.21
United Kingdom	36,800,000	102,000,000	1.01
TOTAL	**593,000,000**	**1,590,000,000**	
% Cost compared to unadjusted costs	94.9%	94.7%	

Notes. Cost estimates are based on general EU-wide degradation levels, and actual Member State costs would differ according to national ecosystem degradation levels. Costs cannot be estimated for Greece due to inadequate CLC data. *Based on 2010 comparative price levels according to Eurostat http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tec00120

Figure 7-3 Total average additional annual maintenance costs in 2020 (million €) for each Member State, with costs adjusted according to variation in Member State prices



The results provided in Tables 7-15 and 7-16 and Figure 7-4 indicate the estimated annual additional costs of restoring each ecosystem under each of the Target 2 **restoration** disaggregation scenarios described above.

Under **Scenario A**, in which 15% of the degraded area of each ecosystem is restored (ie the default scenario used to calculate costs for each ecosystem in chapter 6), the total average annual costs to 2020 range from €7.79 to 10.9 billion. Despite the even percentage restoration allocation across all ecosystems the majority of the costs would be associated with measures for arable ecosystems, as a result of their large area, high levels of degradation and the relatively high cost of restoration measures for them. There would also be substantial costs resulting from restoration of agriculturally improved grasslands and permanent crops. Most other ecosystem costs would be much lower due to the smaller area of their ecosystems, their lower degradation levels and expected measures under the existing initiatives (eg the WFD) measures. Heathland and tundra restoration costs are especially low due to the relatively low pressures on the ecosystem over much of the area and the potential for restoration to occur naturally following alleviation of pressures that may be brought about through regulation rather than funded measures. However, it is clear that to achieve 15% restoration across all ecosystems would require a substantial increase in funding of restoration measures for most ecosystem types.

Scenario B attempts to reduce the cost of achieving Target 2 by focusing restoration measures on ecosystem that have the lowest unit cost for restoration. Under this scenario, 50% of the degraded area is restored in forests, heathland and tundra, mires, lakes and rivers (other than components covered by the WFD) and saltmarshes. This would result in a substantial reduction in the costs of restoration, with the total coming down to €506 million to €1.75 billion. In this scenario the largest proportion of the costs would be related to forest restoration measures, but there would also be significant restoration costs related to semi-natural grasslands and mires.

Scenario C aims to maximize biodiversity conservation benefits, by focusing on the restoration of ecosystems that contain a high proportion of habitat types that are of Community interest and therefore the subject of measures under the Habitats Directive. This would result in the restoration of 50% of the degraded areas of heathland and tundra, mires, dunes and beaches and saltmarsh. There would also be substantial restoration of all other ecosystem types other than arable, permanent crops and improved grassland; restoration would be limited to 6.5% of the degraded area for these latter three ecosystem types.

Under this scenario the total cost of restoration would range from €5.16 billion to 7.87 billion. This is a substantial reduction on Scenario A, and also significantly lower than Scenarios D and E (see below), but much higher than the low cost scenario B. Despite the low percentage allocation to arable, permanent crops and improved grassland ecosystems, they still make up the predominant proportion of the costs due to their large area etc.

Scenario D aims to maximize ecosystem services, by focusing on the restoration of ecosystems that provide the highest overall ecosystem service benefits. This would result in a focus on forests (30% restored), 18% restoration of semi-natural grasslands, heathland and tundra, and mires; 9% restoration of remaining agricultural ecosystems, lakes and rivers and inland marshes. Restoration on all other ecosystems would be limited to 3%.

The costs of restoration under this scenario would range from €5.87 billion to 8.59 billion. As with all scenarios, other than the low cost scenario, the majority of the costs would be associated with agricultural ecosystems, especially arable ecosystems. A significant amount of the costs would also relate to forest ecosystems.

Scenario E attempts to provide a balanced set of ecosystem restoration targets that takes into account the biodiversity conservation importance of ecosystems (as assessed under Scenario C) and ecosystem services (as assessed for Scenario D). Under this scenario the percentage allocation is relatively uniform, with 40% targets for heathland and tundra and mires, and 20% allocation for all other ecosystems, other than arable, permanent crops, improved grassland and sclerophyllous vegetation, which have a 10% target.

The costs of this scenario are slightly higher than Scenario D, ranging from €6.20 billion to 8.97 billion. The cost distribution amongst ecosystem types is very similar to that under Scenario D, although the costs associated with forests are lower

Table 7-17 provides estimates of the national average annual additional costs of restoring ecosystems according to each of the scenarios described above. Table 7-18 and Figure 7-5 then provides adjusted estimates of these costs in relation to variations in purchasing power amongst the countries. As for maintenance costs, discussed above, the variations in costs primarily reflect the area of the countries and their ecosystems (as it is not possible to take into account national degradation levels in each ecosystem type). As a result the highest costs under all scenarios mostly occur within France and Spain (with some scenarios costing over €1 billion) followed by Germany, Italy, Poland, Sweden and the United Kingdom, where most scenario costs range between €400 and 800 million. The average annual costs for Finland, Romania and Sweden are in the region of €200 million. Costs for all other countries are under €100 million for most scenarios.

Table 7-15: Total additional annual costs in 2020 of restoring each ecosystem according to each scenario

Scenario	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
Degradation level	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Arable ecosystems including temporary grasslands	5,960,000,000	6,920,000,000	-	-	3,680,000,000	4,100,000,000	4,350,000,000	4,930,000,000	4,620,000,000	5,261,000,000
Permanent crops	642,000,000	1,150,000,000	-	-	411,000,000	829,000,000	479,000,000	923,000,000	506,000,000	960,000,000
Agriculturally improved permanent grasslands	918,000,000	1,380,000,000	-	-	590,000,000	984,000,000	687,000,000	1,100,000,000	725,000,000	1,150,000,000
Natural and semi-natural grasslands	54,300,000	506,000,000	38,400,000	440,000,000	109,000,000	731,000,000	65,200,000	551,000,000	72,400,000	581,000,000
Forest	140,000,000	519,000,000	390,000,000	1,140,000,000	175,000,000	608,000,000	247,000,000	785,000,000	175,000,000	608,000,000
Heathland and tundra	199,000	1,910,000	557,000	5,230,000	557,000	5,230,000	229,000	2,200,000	455,000	4,280,000
Sclerophyllous vegetation	48,400,000	317,000,000	-	-	96,800,000	420,000,000	9,680,000	236,000,000	32,300,000	283,000,000
Mires (bogs and fens)	9,360,000	28,000,000	49,900,000	91,900,000	49,900,000	91,900,000	12,800,000	33,500,000	38,300,000	73,600,000
Inland marshes	9,750,000	13,600,000	-	-	11,000,000	18,600,000	9,240,000	11,500,000	10,200,000	15,300,000
Lakes and rivers	4,070,000	18,100,000	13,600,000	60,400,000	10,800,000	48,300,000	2,440,000	10,900,000	5,420,000	24,200,000
Dunes & beaches	5,870,000	8,030,000	-	-	14,200,000	21,400,000	3,010,000	3,440,000	7,060,000	9,940,000
Salt marshes	4,250,000	4,470,000	14,200,000	14,900,000	14,200,000	14,900,000	849,000	894,000	5,660,000	5,960,000
Total	**7,790,000,000**	**10,900,000,000**	**506,000,000**	**1,750,000,000**	**5,160,000,000**	**7,870,000,000**	**5,870,000,000**	**8,590,000,000**	**6,200,000,000**	**8,970,000,000**

Table 7-16: Percentage of total restoration costs comprised by each ecosystem under each scenario

Colour shading reflects the proportion of the total cost attributed to each ecosystem type

Scenario	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
Degradation level	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Arable ecosystems including temporary grasslands	76.4%	63.7%	0.0%	0.0%	71.3%	52.1%	74.2%	57.4%	74.5%	58.6%
Permanent crops	8.2%	10.6%	0.0%	0.0%	8.0%	10.5%	8.2%	10.7%	8.2%	10.7%
Agriculturally improved permanent grasslands	11.8%	12.7%	0.0%	0.0%	11.4%	12.5%	11.7%	12.8%	11.7%	12.8%
Natural and semi-natural grasslands	0.7%	4.7%	7.6%	25.1%	2.1%	9.3%	1.1%	6.4%	1.2%	6.5%
Forest	1.8%	4.8%	77.0%	65.1%	3.4%	7.7%	4.2%	9.1%	2.8%	6.8%
Heathland and tundra	0.0%	0.0%	0.1%	0.3%	0.0%	0.1%	0.0%	0.0%	0.0%	0.0%
Sclerophyllous vegetation	0.6%	2.9%	0.0%	0.0%	1.9%	5.3%	0.2%	2.7%	0.5%	3.2%
Mires (bogs and fens)	0.1%	0.3%	9.9%	5.2%	1.0%	1.2%	0.2%	0.4%	0.6%	0.8%
Inland marshes	0.1%	0.1%	0.0%	0.0%	0.2%	0.2%	0.2%	0.1%	0.2%	0.2%
Lakes and rivers	0.1%	0.2%	2.7%	3.4%	0.2%	0.6%	0.0%	0.1%	0.1%	0.3%
Dunes & beaches	0.1%	0.1%	0.0%	0.0%	0.3%	0.3%	0.1%	0.0%	0.1%	0.1%
Salt marshes	0.1%	0.0%	2.8%	0.9%	0.3%	0.2%	0.0%	0.0%	0.1%	0.1%

Figure 7-4 Total additional annual costs (million Euros) in 2020 of restoring each ecosystem according to each scenario

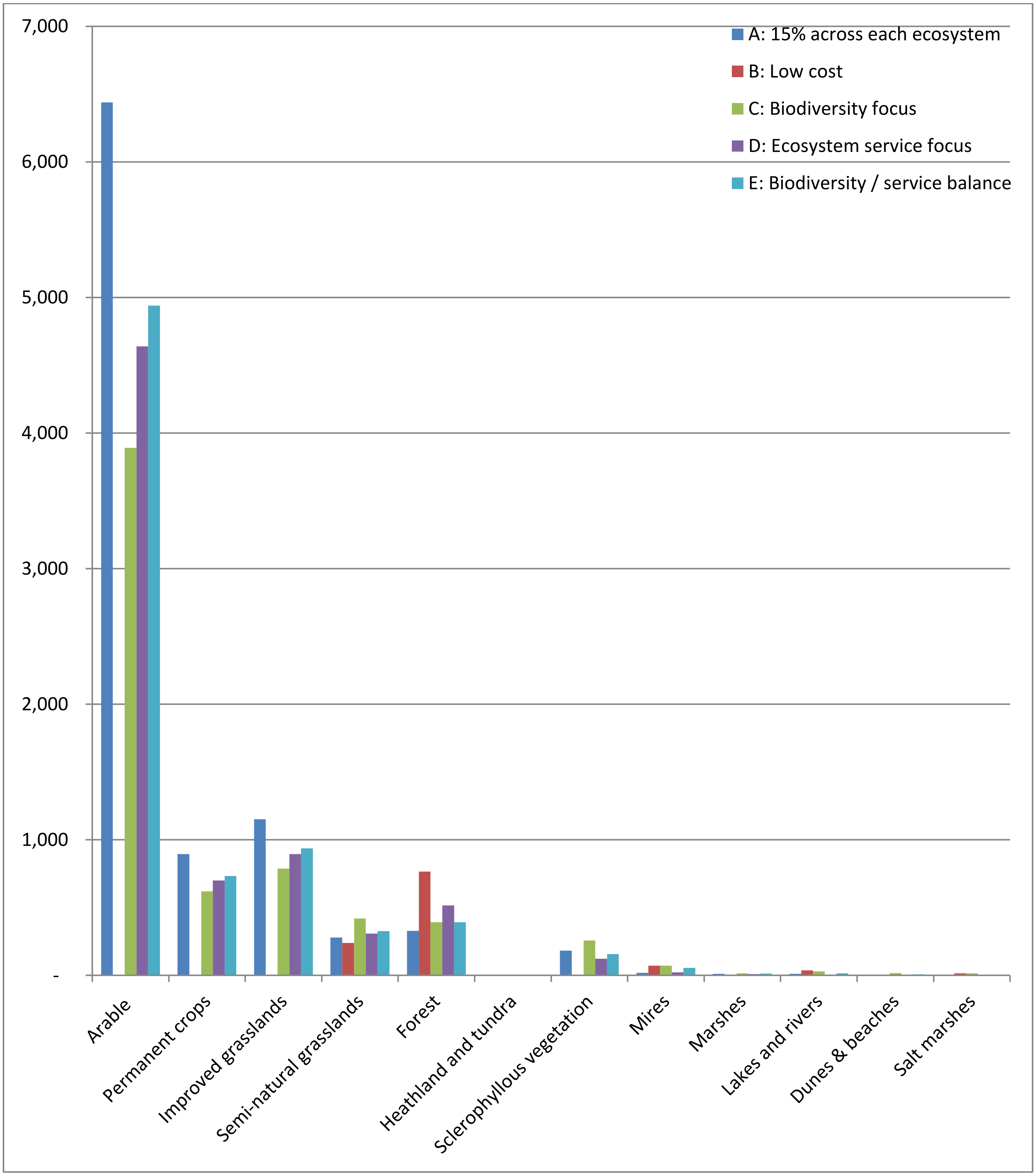


Table 7-17: Total additional annual restoration costs (€) in 2020 for each Member State according to each scenario

	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Austria	86,100,000	134,000,000	11,600,000	48,500,000	59,200,000	109,000,000	67,900,000	116,000,000	70,000,000	117,000,000
Belgium	64,000,000	85,000,000	2,030,000	6,930,000	40,800,000	56,500,000	47,800,000	65,300,000	50,300,000	68,100,000
Bulgaria	230,000,000	304,000,000	12,500,000	49,400,000	148,000,000	210,000,000	173,000,000	238,000,000	18000 ,000	247,000,000
Cyprus	19,900,000	31,400,000	646,000	2,320,000	14,200,000	25,400,000	14,300,000	23,800,000	15,500,000	25,800,000
Czech Republic	173,000,000	224,000,000	8,330,000	32,400,000	111,000,000	152,000,000	130,000,000	174,000,000	136,00,000	181,000,000
Denmark	152,000,000	182,000,000	3,020,000	6,580,000	96,300,000	113,000,000	111,000,000	132,000,000	119,000,00	141,000,000
Estonia	54,100,000	77,200,000	7,930,000	25,400,000	37,100,000	59,100,000	42,500,000	65,100,000	44,200,000	66,400,000
Finland	139,000,000	237,000,000	82,100,000	232,000,000	117,000,000	228,000,000	127,000,000	242,000,000	129,000,000	236,000,000
France	1,290,000,000	1,750,000,000	49,400,000	168,000,000	831,000,000	1,200,000,000	963,000,000	1,360,000,000	1,010,000,000	1,420,000,000
Germany	878,000,000	1,130,000,000	31,900,000	107,000,000	558,000,000	740,000,000	655,000,000	860,000,000	690,000,000	898,000,000
Greece*	?	?	?	?	?	?	?	?	?	?
Hungary	274,000,000	344,000,000	6,310,000	26,200,000	173,000,000	224,000,000	203,000,000	258,000,000	215,000,000	272,000,000
Ireland	152,000,000	228,000,000	9,430,000	24,400,000	105,000,000	175,000,000	115,000,000	183,000,000	125,000,000	197,000,000
Italy	654,000,000	936,000,000	28,900,000	109,000,000	432,000,000	676,000,000	487,000,000	730,000,000	516,000,000	766,000,000
Latvia	82,300,000	123,000,000	10,600,000	40,300,000	56,500,000	97,900,000	64,300,000	105,000,000	67,000,000	107,000,000
Lithuania	161,000,000	206,000,000	6,370,000	19,800,000	102,000,000	135,000,000	121,000,000	158,000,000	127,000,000	164,000,000
Luxemburg	4,970,000	6,870,000	265,000	777,000	3,180,000	4,680,000	3,760,000	5,460,000	3,930,000	5,610,000

Costs of implementing Target 2 of the EU Biodiversity Strategy

	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Malta	411,000	1,150,000	11,300	80,500	408,000	1,250,000	254,000	886,000	314,000	1,010,000
Netherlands	89,200,000	121,000,000	2,040,000	6,720,000	57,900,000	83,600,000	66,200,000	92,700,000	70,400,000	98,100,000
Poland	825,000,000	1,040,000,000	28,100,000	97,500,000	522,000,000	673,000,000	615,000,000	787,000,000	647,000,000	823,000,000
Portugal	149,000,000	237,000,000	10,500,000	42,900,000	102,000,000	185,000,000	113,000,000	193,000,000	119,000,000	201,000,000
Romania	497,000,000	671,000,000	24,500,000	107,000,000	323,000,000	475,000,000	374,000,000	527,000,000	393,000,000	550,000,000
Slovakia	94,800,000	123,000,000	6,150,000	22,000,000	60,900,000	83,800,000	71,700,000	96,900,000	74,900,000	99,900,000
Slovenia	18,900,000	31,100,000	3,460,000	12,400,000	13,300,000	25,600,000	15,100,00	27,500,000	15,500,000	27,500,000
Spain	990,000,000	1,600,000,000	41,500,000	195,000,000	692,000,000	1,290,000,000	726,000,000	1,260,000,000	780,000,000	1,340,000,000
Sweden	198,000,000	322,000,000	99,000,000	284,000,000	161000,000	302,000,000	175,000,000	318,000,000	179,000,000	315,000,000
United Kingdom	515,000,000	718,000,000	19,700,000	86,100,000	342,000,000	528,000,000	384,000,000	562,500,000	412,000,000	599,000,000
TOTAL	**7,790,000,000**	**10,900,000,000**	**506,000,000**	**1,750,000,000**	**5,160,000,000**	**7,850,000,000**	**5,870,000,000**	**8,580,000,000**	**6,200,000,000**	**8,970,000,000**

Note. Based on general EU-wide degradation levels. *No estimates are provided for Greece due to inadequate CLC data.

Table 7-18: Total additional annual restoration costs in 2020 for each Member State according to each scenario, with costs adjusted according to variation in Member State prices

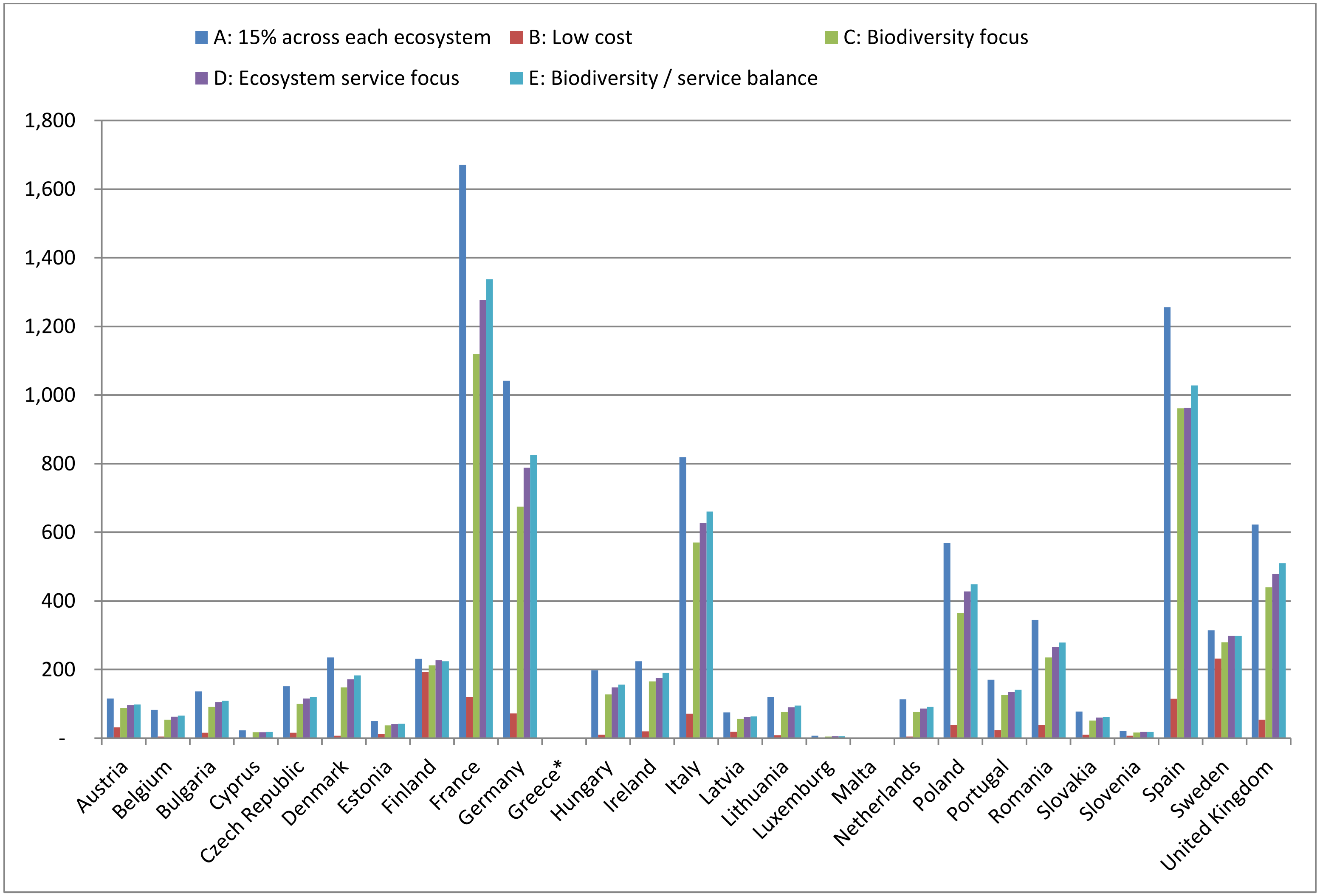
Member State costs	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance		Price adjustme nt
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	
Austria	90,400,000	141,000,000	12,200,000	50,900,000	62,100,000	114,000,000	71,300,000	122,000,000	73,500,000	123,000,000	1.05
Belgium	71,100,000	94,100,000	2,260,000	7,690,000	45,200,000	62,800,000	53,100,000	72,500,000	55,900,000	75,600,000	1.11
Bulgaria	117,000,000	155,000,000	6,360,000	25,200,000	75,500,000	107,000,000	88,300,000	121,000,000	92,500,000	126,000,000	0.51
Cyprus	17,800,000	28,100,000	578,000	2,080,000	12,700,000	22,700,000	12,800,000	21,300,000	13,900,000	23,100,000	0.90
Czech Republic	131,000,000	171,000,000	6,330,000	24,600,000	84,200,000	115,000,000	98,700,000	132,000,000	104,000,000	137,000,000	0.76
Denmark	214,000,000	257,000,000	4,260,000	9,270,000	136,000,000	160,000,000	157,000,000	186,000,000	167,000,000	199,000,000	1.41
Estonia	41,300,000	58,900,000	6,050,000	19,400,000	28,300,000	45,100,000	32,400,000	49,700,000	33,700,000	50,700,000	0.76
Finland	171,000,000	292,000,000	101,000,000	285,000,000	144,000,000	280,000,000	156,000,000	298,000,000	158,000,000	290,000,000	1.23
France	1,420,000,000	1,920,000,000	54,400,000	185,000,000	914,000,000	1,320,000,000	1,060,000,000	1,490,000,000	1,120,000,000	1,560,000,000	1.10
Germany	913,000,000	1,170,000,000	33,200,000	111,000,000	581,000,000	770,000,000	682,000,000	894,000,000	718,000,000	934,000,000	1.04
Greece	?	?	?	?	?	?	?	?	?	?	
Hungary	175,000,000	220,000,000	4,040,000	16,800,000	111,000,000	143,000,000	130,000,000	165,000,000	137,000,000	174,000,000	0.64
Ireland	180,000,000	269,000,000	11,100,000	28,800,000	124,000,000	206,000,000	136,000,000	216,000,000	148,000,000	232,000,000	1.18
Italy	674,000,000	964,000,000	29,800,000	112,000,000	445,000,000	696,000,000	502, ,	752,000,000	532,000,000	789,000,000	1.03
Latvia	60,100,000	90,000,000	7,710,000	29,400,000	41,300,000	71,200,000	47,000,000	76,300,000	48,900,000	78,100,000	0.73
Lithuania	105,000,000	134,000,000	4,140,000	12,900,000	66,600,000	87,600,000	78,400,000	102,000,000	82,400,000	107,000,000	0.65
Luxemburg	6,060,000	8,390,000	323,000	948,000	3,880,000	5,710,000	4,590,000	6,660,000	4,790,000	6,840,000	1.22

Costs of implementing Target 2 of the EU Biodiversity Strategy

Member State costs	A: 15% across each ecosystem		B: Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance		Price adjustment
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	
Malta	317,000	884,000	8,680	62,000	314,000	963,000	196,000	682,000	242,000	773,000	0.77
Netherlands	96,400,000	131,000,000	2,200,000	7,250,000	62,500,000	90,300,000	71,500,000	100,000,000	76,000,000	106,000,000	1.08
Poland	503,000,000	633,000,000	17,100,000	59,500,000	319,000,000	411,000,000	375,000,000	480,000,000	395,000,000	502,000,000	0.61
Portugal	131,000,000	208,000,000	9,270,000	37,700,000	89,500,000	162,000,000	99,300,000	170,000,000	105,000,000	177,000,000	0.88
Romania	293,000,000	396,000,000	14,500,000	63,200,000	191,000,000	280,000,000	221,000,000	311,000,000	232,000,000	325,000,000	0.59
Slovakia	67,300,000	87,600,000	4,370,000	15,600,000	43,300,000	59,500,000	50,900,000	68,800,000	53,200,000	70,900,000	0.71
Slovenia	16,200,000	26,800,000	2,980,000	10,700,000	11,400,000	22,100,000	13,000,000	23,700,000	13,300,000	23,600,000	0.86
Spain	960,000,000	1,550,000,000	40,300,000	189,000,000	671,000,000	1,250,000,000	704,000,000	1,220,000,000	757,000,000	1,300,000,000	0.97
Sweden	239,000,000	390,000,000	120,000,000	344,000,000	194,000,000	365,000,000	216,000,000	385,000,000	217,000,000	381,000,000	1.21
United Kingdom	520,000,000	725,000,000	19,900,000	86,900,000	346,000,000	533,000,000	388,000,000	568,000,000	416,000,000	605,000,000	1.01
TOTAL	**7,210,000,000**	**10,125,838,838**	**514,000,000**	**1,730,000,000**	**4,800,000,000**	**7,390,000,000**	**5,440,000,000**	**8,040,000,000**	**5,750,000,000**	**8,400,000,000**	
% Cost compared to unadjusted costs	**92.6%**	**93.3%**	**101.5%**	**99.0%**	**93.1%**	**94.0%**	**92.8%**	**93.7%**	**92.8%**	**93.7%**	

Note: Based on general EU-wide degradation levels. No estimates are provided for Greece due to inadequate CLC data. * Based on 2010 comparative price levels according to Eurostat http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tec00120

Figure 7-5 Average total additional annual restoration costs (million €) in 2020 for each Member State according to each scenario, with costs adjusted according to variation in Member State prices



The results provided in Tables 7-19 and 7-20 and Figure 7-6 indicate the estimated annual additional costs of **re-creating** each ecosystem under each of the Target 2 disaggregation scenarios described above. When interpreting these results it should be borne in mind that there are no re-creation targets for arable, permanent crops or improved grasslands (as these artificial ecosystems do not merit re-creation), forests (because there has been a net increase in their area in recent decades) and mires, lakes and rivers (because they cannot normally be feasibly re-created with reasonable costs).

Under **Scenario A**, 15% of the area of each ecosystem that was converted since 2000, and is expected to be converted to 2020, would be re-created by 2020. The cost of re-creation under this scenario would range from €66 to 117 million. The majority of the costs would result from the re-creation of semi-natural grasslands and to a lesser extent sclerophyllous vegetation. Most other ecosystem re-creation costs would be much lower due to the smaller area of their ecosystems and low rates of conversion.

Scenario B attempts to reduce the cost of achieving Target 2 by focusing on ecosystems that have the lowest unit cost for re-creation. Under this scenario, re-creation focusses on heathland and tundra, dunes and beaches and saltmarsh, with 50% of the converted area of these ecosystems re-created. In addition 36% of the converted area of sclerophyllous vegetation and inland marshes would be re-created. This scenario would result in a reduction in the costs of ecosystem re-creation, with the total coming down to €54 to 82 million. In this scenario, due to its relatively large area, the greatest proportion of the costs would be related to the re-creation of sclerophyllous vegetation.

Scenario C aims to maximize biodiversity conservation benefits, by focusing on the re-creation of ecosystems that contain a high proportion of habitat types that are the subject of measures under the Habitats Directive. This would result in the re-creation of 35% of the converted areas of heathland and tundra, dunes and beaches and saltmarsh. For all other ecosystems the re-creation target is 14.6% of the re-created area. Under this scenario the total additional re-creation cost would range from €64 to 115 million. Most of the cost would be associated with semi-natural grasslands and sclerophyllous vegetation due to the extensive area of these ecosystems.

Scenario D aims to maximize ecosystem services, by focusing on the re-creation of ecosystems that provide the highest overall ecosystem service benefits. This would result in a focus on semi-natural grasslands, heathlands and tundra (21% converted re-created), 12% re-creation of inland marshes and 5% re-creation of other ecosystems. The costs of re-creation under this scenario would range from €69 to 129 million. As under Scenario C the majority of the costs would result from the re-creation of semi-natural grasslands and sclerophyllous vegetation.

Scenario E attempts to provide a balanced set of ecosystem re-creation targets that takes into account the biodiversity conservation importance of ecosystems (as assessed under Scenario C) and ecosystem services (as assessed for Scenario D). Under this scenario the re-creation target is 38% for heathland and tundra, 18% for all other ecosystem types other than sclerophyllous vegetation, which has a 9% target. The costs of this scenario are similar to Scenario D, ranging from €66 to 122 million. The cost distribution amongst ecosystem types is also very similar to that under Scenario D, although the costs of restoring sclerophyllous vegetation is higher.

Table 7-21 provides estimates of the national average annual additional costs of re-creating ecosystems according to each of the scenarios described above. Table 7-22 and Figure 7-7 then provide adjusted estimates of these costs in relation to variations in purchasing power amongst the countries. As for maintenance and restoration costs, the variations in costs primarily reflect the area of the countries and their ecosystems. It should be remembered that it is not possible to take into account variations in conversion rates amongst counties (and costs are therefore based on average EU conversion rates).

Under all scenarios it is clear that the highest average additional annual re-creation costs to 2020 would occur in Spain (between €25 and 45 million), primarily as result of its large area of sclerophyllous vegetation and semi-natural grasslands. Greece also has a large area of these ecosystems and would also probably incur high costs as a result, but CLC data are inadequate to include Greece in this is Member State analysis, and this cannot therefore be confirmed. Other countries with re-creation costs that are above €5 million per year are France, Italy and the United Kingdom.

Table 7-19: Total additional annual costs (€) in 2020 of re-recreating each ecosystem according to each scenario

Scenario	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
Degradation level	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Natural and semi-natural grasslands	43,400,000	83,500,000	-	-	42,200,000	81,300,000	60,700,000	117,000,000	52,000,000	100,000,000
Heathland and tundra	109,000	387,000	363,000	1,290,000	254,000	903,000	153,000	542,000	254,000	903,000
Sclerophyllous vegetation	21,500,000	32,300,000	51,500,000	77,200,000	20,900,000	31,400,000	7,170,000	10,800,000	12,900,000	19,400,000
Inland marshes	953,000	953,000	2,330,000	2,330,000	928,000	928,000	763,000	763,000	1,140,000	1,140,000
Dunes & beaches	9,740	97,400	32,500	325,000	22,700	227,000	3,250	32,500	11,700	117,000
Salt marshes	17,900	112,000	59,600	372,000	41,700	261,000	5,960	37,200	21,500	134,000
Total	**66,000,000**	**117,000,000**	**54,300,000**	**81,600,000**	**64,400,000**	**115,000,000**	**68,800,000**	**129,000,000**	**66,400,000**	**122,000,000**

NB. There are no re-creation targets for arable, permanent crops, agriculturally improved grasslands, forests, mires, or lakes and rivers.

Table 7-20: Percentage of total re-recreation costs comprised by each ecosystem under each scenario

Colour shading reflects the proportion of the total cost attributed to each ecosystem type

Scenario	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
Ecosystem type	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Natural and semi-natural grasslands	66%	71%	0%	0%	66%	71%	88%	91%	78%	82%
Heathland and tundra	0%	0%	1%	2%	0%	1%	0%	0%	0%	1%
Sclerophyllous vegetation	33%	28%	95%	95%	33%	27%	10%	8%	19%	16%
Inland marshes	1%	1%	4%	3%	1%	1%	1%	1%	2%	1%
Dunes & beaches	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Salt marshes	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

NB. There are no re-creation targets for arable, permanent crops, agriculturally improved grasslands, forests, mires, or lakes and rivers.

Figure 7-6 Total average additional annual re-creation costs in 2020 (million €) for each ecosystem according to each scenario

NB. There are no re-creation targets for arable, permanent crops, agriculturally improved grasslands, forests, mires, or lakes and rivers.

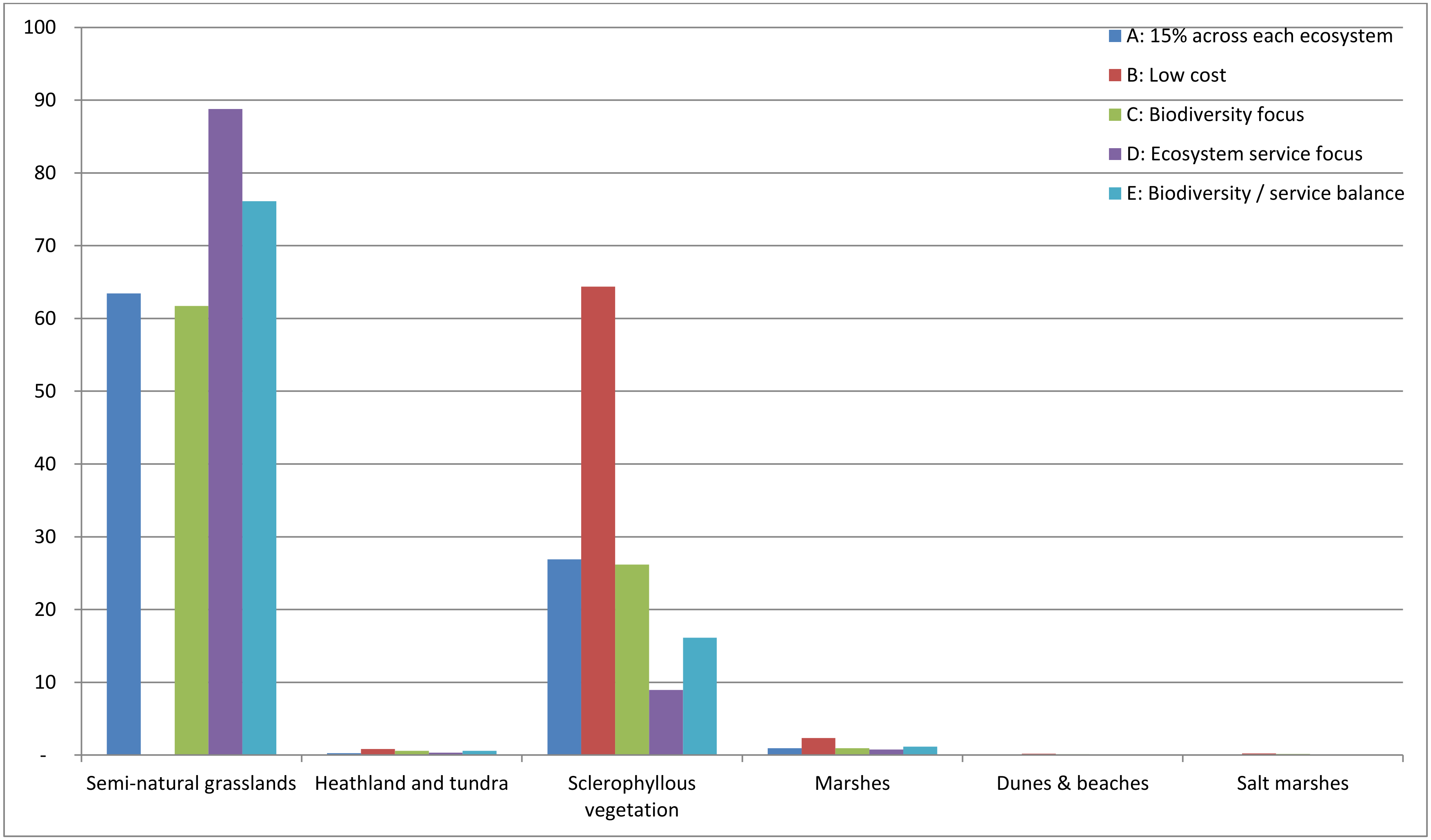


Table 7-21: Total additional annual re-recreation costs (€) in 2020 for each Member State according to each scenario

	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Austria	1,920,000	3,680,000	55,800	81,300	1,870,000	3,600,000	2,680,000	5,150,000	2,310,000	4,430,000
Belgium	138,000	264,000	7,090	10,500	135,000	259,000	191,000	367,000	166,000	317,000
Bulgaria	1,690,000	3,260,000	21,500	27,400	1,650,000	3,170,000	2,370,000	4,560,000	2,030,000	3,910,000
Cyprus	503,000	790,000	1,010,000	1,530,000	490,000	772,000	253,000	428,000	350,000	568,000
Czech Republic	1,060,000	2,040,000	13,300	13,500	1,035,000	1,990,000	1,490,000	2,860,000	1,280,000	2,450,000
Denmark	96,400	172,000	73,800	120,000	97,600	190,000	117,000	211,000	116,000	210,000
Estonia	428,000	760,000	172,000	178,000	417,000	743,000	557,000	1,020,000	513,000	913,000
Finland	288,000	547,000	80,300	149,000	290,000	569,000	388,000	750,000	352,000	681,000
France	5,530,000	10,000,000	3,720,000	5,640,000	5,400,000	9,810,000	6,130,000	11,600,000	5,760,000	10,700,000
Germany	2,000,000	3,820,000	124,000	160,000	1,950,000	3,740,000	2,780,000	5,310,000	2,410,000	4,590,000
Greece*	?	?	?	?	?	?	?	?	?	?
Hungary	1,060000	1,990,000	168,000	168,000	1,040,000	1,930,000	1,500,000	2,740,000	1,280,000	2,380,000
Ireland	629,000	1,200,000	41,900	66,400	615,000	1,180,000	872,000	1,670,000	757,000	1,450,000
Italy	6,290,000	11,000,000	6,420,000	9,730,000	6,130,000	10,800,000	5,960,000	11,100,000	5,960,000	10,800,000
Latvia	1,370,000	2,610,000	52,000	56,170	1,330,000	2,540,000	1,900,000	3,600,000	1,640,000	3,130,000
Lithuania	177,000	325,000	42,700	45,500	172,000	318,000	237,000	444,000	212,000	391,000
Luxemburg	-	-	-	-	-	-	-	-	-	-
Malta	51,200	79,400	107,000	161,000	49,800	77,300	23,800	39,400	34,500	55,000
Netherlands	167,000	300,000	81,700	109,000	165,000	305,000	214,000	393,000	201,000	362,000
Poland	2,010,000	3,790,000	219,000	224,000	1,960,000	3,690,000	2,760,000	5,250,000	2,420,000	4,550,000
Portugal	2,490,000	4,510,000	1,720,000	2,640,000	2,430,000	4,430,000	2,730000	5,170,000	2,570,000	4,800,000
Romania	4,950,000	9,270,000	746,000	775,000	4,830,000	9,040,000	6,750,000	12,800,000	5,950,000	11,100,000
Slovakia	530,000	1,020,000	6,550	8,140	516,000	992,000	740,000	1,420,000	636,000	1,220,000
Slovenia	376,000	694,000	172,000	259,000	367,000	677,000	451,000	858,000	410,000	770,000
Spain	25,300,000	42,183,000	36,800,000	55,300,000	24,600,000	41,200,000	19,100,000	34,500,000	21,200,000	36,900,000

Costs of implementing Target 2 of the EU Biodiversity Strategy

	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Sweden	906,000	1,750,000	349,000	734,000	936,000	1,890,000	1,200,000	2,350,000	1,110,000	2,220,000
United Kingdom	5,180,000	10,000,000	141,000	473,000	5,080,000	9,940,000	7,230,000	14,000,000	6,240,000	12,100,000
TOTAL	**65,100,000**	**116,000,000**	**52,300,000**	**78,600,000**	**63,600,000**	**114,000,000**	**68,500,000**	**129,000,000**	**65,900,000**	**121,000,000**

Note: Based on general EU-wide conversion rates. No estimates are provided for Greece due to inadequate CLC data

Table 7-22: Total additional annual re-recreation costs (€) in 2020 for each Member State according to each scenario, with costs adjusted according to variation in Member State prices

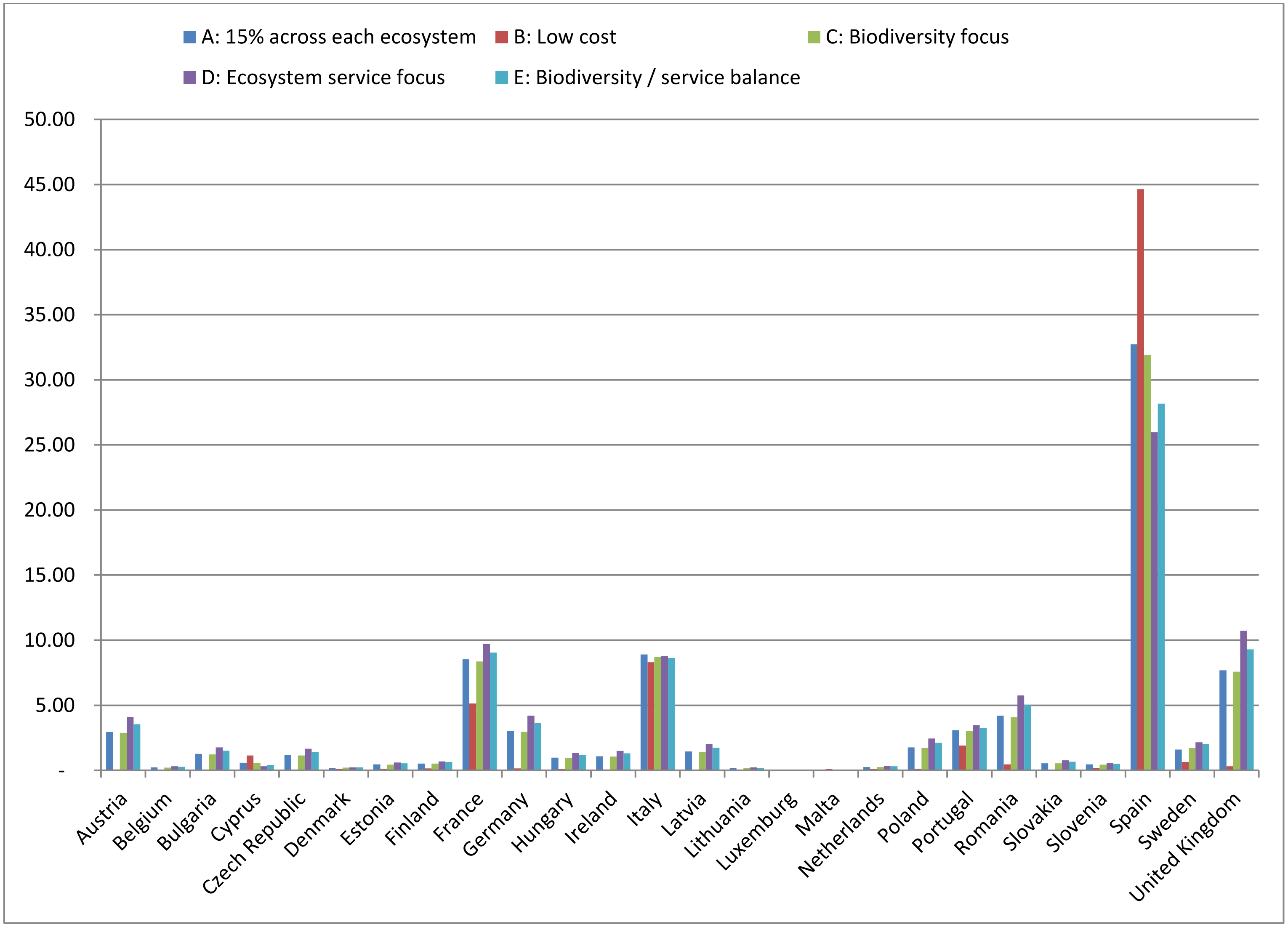
Member State costs	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance		Price adjustment
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	
Austria	2,020,000	3,870,000	58,600	85,400	1,970,000	3,780,000	2,809,000	5,400,000	2,422,000	4,650,000	1.05
Belgium	153,000	293,000	7,860	11,700	149,000	287,000	212,000	408,000	184,000	352,000	1.11
Bulgaria	864,000	1,660,000	11,000	14,000	842,000	1,620,000	1,210,000	2,320,000	1,040,000	1,990,000	0.51
Cyprus	450,000	707,000	907,000	1,370,000	439,000	691,000	226,000	383,000	313,000	508,000	0.90
Czech Republic	808,000	1,550,000	10,100	10,200	786,000	1,510,000	1,130,000	2,170,000	970,000	1,860,000	0.76
Denmark	136,000	243,000	104,000	169,000	138,000	269,000	165,000	297,000	164,000	296,000	1.41
Estonia	326,000	580,000	131,000	136,000	318,000	567,000	425,000	779,000	392,000	697,000	0.76
Finland	354,000	672,000	98,800	183,000	356,000	700,000	477,000	917,000	433,000	838,000	1.23
France	6,080,000	11,000,000	4,090,000	6,200,000	5,940,000	10,800,000	6,740,000	12,700,000	6,340,000	11,800,000	1.10
Germany	2,080,000	3,980,000	129,000	167,000	2,030,000	3,890,000	2,890,000	5,530,000	2,500,000	4,780,000	1.04
Greece*	?	?	?	?	?	?	?	?	?	?	
Hungary	681,000	1,270,000	108,000	108,000	663,000	1,240,000	927,000	1,750,000	818,000	1,530,000	0.64
Ireland	743,000	1,420,000	49,400	78,400	726,000	1,400,000	1,030,000	1,970,000	893,000	1,710,000	1.18
Italy	6,480,000	11,300,000	6,610,000	10,000,000	6,310,000	11,100,000	6,140,000	11,400,000	6,130,000	11,100,000	1.03
Latvia	997,000	1,910,000	37,900	41,000	970,000	1,860,000	1,390,000	2,660,000	1,200,000	2,290,000	0.73
Lithuania	115,000	211,000	27,800	29,500	112,000	207,000	154,000	289,000	138,000	254,000	0.65
Luxemburg	-	-	-	-	-	-	-	-	-	-	1.22
Malta	39,400	61,200	82,700	124,000	38,400	59,500	18,300	30,300	26,600	42,300	0.77
Netherlands	181,000	324,000	88,200	118,000	178,000	330,000	231,000	424,000	217,000	391,000	1.08
Poland	1,230,000	2,310,000	133,000	137,000	1,190,000	2,250,000	1,690,000	3,210,000	1,470,000	2,780,000	0.61
Portugal	2,190,000	3,970,000	1,510,000	2,320,000	2,140,000	3,900,000	2,400,000	4,550,000	2,260,000	4,220,000	0.88
Romania	2,920,000	5,470,000	440,000	458,000	2,850,000	5,330,000	3,990,000	7,540,000	3,510,000	6,560,000	0.59
Slovakia	376,000	723,000	4,650	5,770	366,000	704,000	525,000	1,010,000	452,000	868,000	0.71

Costs of implementing Target 2 of the EU Biodiversity Strategy

Member State costs	A: 15% across each ecosystem		B. Low cost		C: Biodiversity focus		D: Ecosystem service focus		E: Biodiversity / service balance		Price adjustment
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	
Slovenia	324,000	596,000	148,000	222,000	315,000	582,000	388,000	738,000	353,000	663,000	0.86
Spain	24,500,000	40,900,000	35,700,000	53,600,000	23,900,000	39,900,000	18,500,000	33,500,000	20,500,000	35,800,000	0.97
Sweden	1,100,000	2,110,000	423,000	888,000	1,130,000	2,290,000	1,450,000	2,850,000	1,340,000	2,690,000	1.21
United Kingdom	5,230,000	10,100,000	142,000	477,000	5,130,000	10,000,000	7,310,000	14,100,000	6,310,000	12,300,000	1.01
TOTAL	**60,400,000**	**107,000,000**	**51,000,000**	**77,000,000**	**59,000,000**	**105,000,000**	**62,400,000**	**117,000,000**	**60,400,000**	**111,000,000**	
% Cost compared to unadjusted costs	92.7%	92.4%	97.5%	97.9%	92.8%	92.5%	91.1%	91.0%	91.7%	91.6%	

Note. Based on general EU-wide conversion rates. No estimates are provided for Greece due to inadequate CLC data *Based on 2010 comparative price levels according to Eurostat http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tec00120

Figure 7-7 Average total additional annual re-recreation costs (million €) in 2020 for each Member State according to each scenario, with costs adjusted according to variation in Member State prices



8 REVIEW OF EXISTING FINANCING MEANS TO REACH TARGET 2

8.1 Introduction

This section provides an overview of the various means by which the restoration of ecosystems can be funded in the EU, through EU and national level funding. The review considers public funding instruments first, as some of these are the most important funders of habitat restoration in the EU (especially the CAP). The sources of private financing and their potential for supporting ecosystem restoration are examined separately in chapter 9, alongside analysis and examples of how public and private sources can be used together. Chapter 10 provides an overview and summary of financing of actions under target 2.

The current chapter is largely based on a review of available published information, including policy documents such as Commission Communications and Staff Working Papers. However, at the time of preparing the initial draft of this report (November/early December 2012) there was uncertainty over the likely outcome of negotiations concerning the EU budget and proposed reforms of several key policy areas, including the CAP, Cohesion Policy, Regional Policy and the Common Fisheries Policy, all of which are being influenced by the Europe 2020 agenda. The MFF has since been agreed as has the CAP but the final legal texts and regulation are yet to be published. Therefore, every effort has been made to ensure that the accounts below reflect the most up-to-date published policy proposals and debates. Nevertheless, there remain some details to be ironed out, not least the final drafting and implementation of Rural Development Programmes and how Member States choose to implement certain measures.

8.2 Common Agricultural Policy

This section considers the articles and measures available under the current Common Agricultural Policy (CAP) regulations (2007 – 2013), specifically those that have the potential to support ecosystem restoration. The proposed changes to the structure and content of certain regulations under the CAP are discussed later in this section. Therefore the section concludes with a reflection on the opportunities that may be available under the future CAP (2014 – 2020) and how this could support the delivery of Target 2 of the biodiversity strategy.

8.2.1 Introduction

With over 40 % of the EU-27 managed for agricultural purposes the Common Agricultural Policy (CAP) and its component measures and instruments provides one of the greatest opportunities in terms of both scope and budget to aid in the restoration of ecosystems. Maintaining and enhancing biodiversity has been one of the key priorities to be addressed by environmental measures within the CAP ever since they were introduced in the 1980s. Many CAP measures have the potential to deliver biodiversity benefits but not all measures set this out as their core objective (Poláková et al, 2011).

Under both Pillar 1 and Pillar 2 of the CAP there are a variety of different measures and practices that have the potential to contribute towards ecosystem restoration. These range from the targeted interventions that seek to restore and maintain ecosystems and their functions, to those that help to reduce the pressures on ecosystems that result from agricultural practices. Both types are considered.

The environmental potential of the CAP has been most often examined with a focus on objectives such as water quality, biodiversity and soil functionality. Few policy studies relate specifically to ecosystem restoration potential. Practices that benefit biodiversity can, in general terms, be considered as helping to contribute towards ecosystem health. Therefore we use the term 'biodiversity' as a proxy for ecosystem health throughout the remainder of this section.

8.2.2 CAP measures with the potential to benefit biodiversity

Poláková *et al* (2011) provides a comprehensive review of the different CAP measures that have the potential to provide biodiversity benefits. Building on the work of Cooper *et al* (2009) they identify three types of measures under the CAP, those that have a direct focus on biodiversity, those that have a partial focus, and those with no direct focus on biodiversity but have the potential to provide benefits. These are set out in Annex 2 Tables 1 and 2[118]. Relevant to this study, priorities for water management, climate change and biodiversity all include operations and potential effects that could contribute towards the restoration of ecosystems, and these are set out in Annex 2 Table 3.

The measures, practices, monitoring, advice and support provided through Rural Development Programmes (RDPs) under Pillar 2 of the CAP provide the greatest potential to deliver biodiversity objectives across the EU (see Box 8.1). The role of Pillar 1 measures, most notably cross compliance, is also important to consider. The following sections discuss in more detail the key measures under both Pillars that have the potential to contribute towards target two of the biodiversity strategy and the financial resources devoted to them.

Box 8.1: Structure of rural development programmes in the EU

The rural development policy (as defined in Council Regulation (EC) No 1698/2005) provides a set of tools (measures) from which all Member States can choose and for which they can receive EU financial support to implement integrated Rural Development Programmes. Each axis is implemented through this menu of measures.

Axis 1: to improve the competitiveness of the agricultural and forestry sector including a range of measures that target human and physical capital in the agriculture, food and forestry sectors (promoting knowledge transfer and innovation) and quality production. 10 % minimum funding requirement

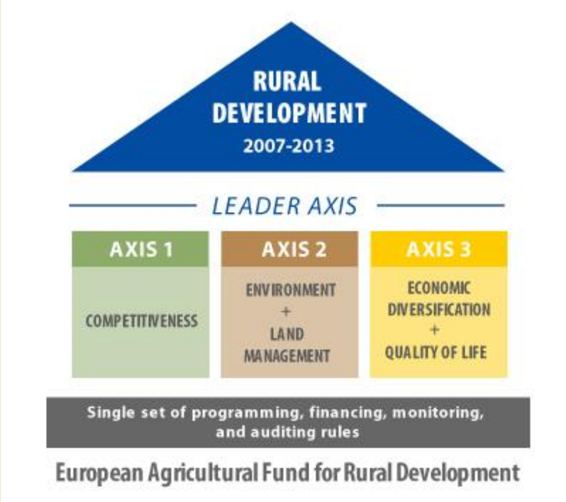
Axis 2: to improve the environment and the countryside, providing measures to protect and enhance natural resources, as well as preserving high value farming and forestry systems and cultural landscapes in Europe's rural areas. 25 % minimum funding requirement

[118] Further information used to supplement this review is taken from Annex II of the European Agricultural Fund for Rural Development (EAFRD) regulation (Council regulation (EC) No 1698/2005), which provides an Indicative list of the types of operations and potential effects, related to specific priorities of the EAFRD (as referred to in points (a) to (f) of Article 16a(1)).

Axis 3: to enhance the quality of life in rural areas and diversification of the rural economy, offering support develop local infrastructure and human capital in rural areas, to improve the conditions for growth and job creation in all sectors and the diversification of economic activities. 10 % minimum funding requirement

Axis 4: based on the Leader experience, introduces possibilities for innovative governance through locally based, bottom-up approaches to rural development. 5 % minimum funding requirement (2.5 % in new Member States)

Funding
To ensure a balanced strategy, a minimum funding for each thematic axis is required to ensure that each programme reflects at least the three main policy objectives. However the percentages are set sufficiently low to leave Member States or regions a high margin of flexibility (55% of EU funding) to emphasize the policy axis they wish in function of their situation and needs.



Source: ENRD http://enrd.ec.europa.eu/policy-in-action/rural-development-policy-overview/axes-and-measures/en/axes-and-measures_en.cfm

8.2.3 Measures under Pillar 2 of the CAP

The RDPs implemented in 2007-2013 period were designed on the basis of a suite of 46 different rural development measures, 23 of which have the potential to deliver biodiversity benefits. The two measures that are essential for restoration activities are the agri-environment measure and the non-productive investment measure, measures 214 and 216 of axis 2 as supported under Articles 39 and 41 of the EAFRD[119] regulation respectively. The forest environment measure (225 supported under Article 47) also has significant potential for restoration activities in woodland and forest areas; however its use across the EU has been relatively limited.

Across the EU agri-environment is the most significant measure within the CAP for delivering biodiversity outcomes, both in terms of the expenditure allocated to the measure under the 2007-13 programming period and because of its compulsory nature. Furthermore it is the only measure that allows support for targeted biodiversity management, which can aid in ecosystem restoration. In the 2007–13 programming period the total budget allocated to this measure amounts to €38 billion (including national co-financing), 23 % of the total Pillar 2 budget (see Table 8-1). It is estimated that the end of 2013 will see approximately 42 million hectares of agricultural land under environmental management, which includes restoration activities, accounting for 24 % of the total Utilised Agricultural Area (UAA) of the EU. The potential to tailor the agri-environment measure allows Member States to introduce schemes and management practices to reflect different bio-physical, climatic, environmental and agronomic conditions and therefore to adapt management options to suit the particular combination of local biodiversity needs which vary enormously even within one region (OECD, 1993, 2010). Examples of successful habitat restoration in the UK

[119] Supported through the EAFRD and in pre-accession countries through the SAPARD (Council Regulation (EC) No 1268/1999) and IPARD (Council Regulation (EC) 1085/2006) regulations.

using support under the agri-environment measures are given in Poláková et al (2011 p.70 Box 9). Restoration actions can include:

- the blocking of drains or ditches to increase water levels;
- the control (increase or decrease) of livestock to maintain sward heights and vegetation structure;
- replanting of vegetation such as trees and shrubs or the sowing of seed mixes;
- and the reduction of inputs such as fertilisers and plant protection products.

The non-productive investments measure is another key measure that is used to secure the delivery of biodiversity benefits, and can be therefore be very helpful for restoration activities, typically in combination with the agri-environment measure. Non-productive investments can be used to enable farmers to undertake capital works on their land, which are not linked to production, for example:

- the blocking of drains or ditches to increase water levels;
- the planting of hedges and erection of fences to control livestock and animal movements.

As a result this measure is particularly effective in aiding in the recreation and restoration of certain biotypes. Although this measure is not compulsory, and also does not command a dedicated share of the Pillar 2 budget, the programmed expenditure across the EU is significant at around €1.1bn.

A further range of measures under Pillar 2 is considered to have a partial impact on biodiversity (see annex 2) and for ecosystem restoration. These include amongst others: advice, training and information measures; LFA and Natura measures; conservation of rural heritage; and LEADER. Advice, training and information measures that help improve the understanding of farmers in relation to ecosystem restoration and how this fits with their agronomic situation can deliver environmental benefits. Farm modernisation measures can help to increase the economic viability of farms, particularly those in remote areas where the risk of abandonment and as such help to prevent the loss of ecosystems dependent on agriculture. Similar benefits are demonstrated through the LEADER approach, which aims to build capacity for added value products by encouraging farmers to identify innovative solutions and use their expertise to complement more formalised policy and science expertise. This may prove particularly useful in the restoration of certain types of ecosystem. Organic farming, marketing of local and quality food, beneficial land management at the landscape scale, development of Natura 2000 management plans, and transnational projects aimed at protected cross-border habitats have also proved helpful to biodiversity goals (Cooper et al., 2006; Beaufoy and Marsden, 2010; Redman, 2010). Although in themselves these measures do not have a direct focus on biodiversity many have been demonstrated to deliver biodiversity benefits, and therefore potential to contribute towards Target 2. In particular these measures have been used successfully in order to provide added value in combination with the agri-environment measure. Combining agri-environment management in particular with the LFA and Natura measures has been demonstrated as particularly effective at delivering both biodiversity and economic benefits in more remote and low productivity areas (Poláková et al, 2011).

8.2.4 Financing

Support for all rural development measures including those with the potential to contribute towards ecosystem restoration is provided through the EAFRD regulation and requires the co-financing by Member States of support granted from the EU. Table 8-1 shows the total programmed expenditure for all rural development measures under the CAP for the 2007 – 2013 programming period. The breakdown between the EAFRD contribution (EU funds) and the total public contribution (including national co-financing) is provided. Table 8-1 gives a graphical representation of the expenditure and clearly highlights the differences in total contributions between measures. By far the largest contribution of public support is provided to the agri-environment measure (214) reflecting both its compulsory nature and the dedicated budget proportion towards axis 2. The following six measures also command significant resources:

- 121 - Modernisation of agricultural holdings
- 123 - Adding value to agricultural and forestry products
- 125 - Infrastructure related to the development and adaptation of agriculture and forestry
- 211 - Natural handicap payments to farmers in mountain areas
- 212 - Payments to farmers in areas with handicaps, other than mountain areas
- 413 - Quality of life/diversification

The funding information provided in Table 8-1 is helpful in showing the distribution of funds across the EAFRD measures; however it is not possible to show the share of this funding devoted to restoration activities for a number of reasons. First, restoration measures are always part of broader biodiversity measures. Second, many of the measures supported through rural development funding provide multiple environmental benefits. For example, environmental land management practices delivered through entry-level agri-environment schemes in the EU are inherently multi-objective (see for example Keenleyside *et al*, 2011). Therefore identifying the cost requirements for a single objective should be treated with caution (see for example Hart *et al*, 2011). Third, it is common throughout the EU for a range of CAP measures to be used together on individual farms, for example cross compliance, agri-environment and farm advice. Across multiple farms it is reasonable to expect a much wider range of measures to be adopted and therefore linking biodiversity outcomes to individual measures at the landscape scale is problematic.

Table 8-1: Total programmed EAFRD and public expenditure per rural development measure in 2007 - 2013 that may include restoration activities

Measure	Description	EAFRD	Total Public
111	***Vocational training and information actions***	***€ 1,088,770,755.00***	***€ 1,796,410,134.92***
112	Setting up of young farmers	€ 2,887,459,093.00	€ 5,034,290,607.97
113	Early retirement	€ 2,853,038,896.00	€ 4,096,489,674.42
114	***Use of advisory services***	***€ 440,116,503.00***	***€ 734,492,349.13***
115	Setting up of management, relief and advisory services	€ 93,521,358.00	€ 154,279,774.35
121	Modernisation of agricultural holdings	€ 10,667,014,207.00	€ 17,311,808,042.72
122	Improvement of the economic value of forests	€ 653,687,055.00	€ 991,593,963.56
123	***Adding value to agricultural and forestry products***	***€ 5,647,323,016.00***	***€ 9,047,691,641.87***
124	Cooperation for development of new products, processes and technologies in the agriculture and food sector and in the forestry sector	€ 349,276,602.00	€ 604,663,151.26
125	Infrastructure related to the development and adaptation of agriculture and forestry	€ 5,129,438,277.00	€ 8,074,187,454.86
126	Restoring agricultural production potential	€ 505,274,802.00	€ 768,184,180.78
131	Meeting standards based on EU legislation	€ 103,920,898.00	€ 165,307,693.75
132	***Participation of farmers in food quality schemes***	***€ 294,073,244.00***	***€ 499,982,345.46***
133	Information and promotion activities	€ 206,366,222.00	€ 378,754,730.45
141	***Semi-subsistence farming***	***€ 993,869,819.00***	***€ 1,282,957,143.31***
142	Producer groups	€ 327,863,144.00	€ 433,429,011.94
143	Providing farm advisory and extension services	€ 131,773,438.00	€ 164,711,091.35
144	Holdings undergoing restructuring due to a reform of a common market organisation	€ 17,030,527.00	€ 29,598,443.93
Total Axis 1		€ 32,389,817,856.00	€ 51,568,831,436.04
211	***Natural handicap payments to farmers in mountain areas***	***€ 6,240,877,766.00***	***€ 10,877,714,794.56***
212	***Payments to farmers in areas with handicaps, other than mountain areas***	***€ 7,241,359,414.00***	***€ 11,874,499,564.68***
213	***Natura 2000 payments and payments linked to Directive 2000/60/EC***	***€ 476,726,824.00***	***€ 777,379,213.01***
214	***Agri-environment payments***	***€ 22,231,273,684.00***	***€ 37,630,390,560.89***
215	Animal welfare payments	€ 543,036,224.00	€ 1,023,089,717.74
216	***Non-productive investments***	***€ 591,086,049.00***	***€ 1,092,995,826.92***
221	First afforestation of agricultural land	€ 2,297,501,976.00	€ 3,407,218,108.92
222	First establishment of agroforestry systems on agricultural land	€ 16,382,490.00	€ 24,877,354.67
223	First afforestation of non-agricultural land	€ 347,805,392.00	€ 528,390,573.31
224	***Natura 2000 payments***	***€ 101,956,083.00***	***€ 144,493,845.92***

225	***Forest-environment payments***	***€ 271,411,253.00***	***€ 423,284,054.51***
226	***Restoring forestry potential and introducing prevention actions***	***€ 1,609,673,680.00***	***€ 2,487,152,363.76***
227	***Non-productive investments***	***€ 808,940,730.00***	***€ 1,303,452,866.39***
Total Axis 2		€ 42,778,031,565.00	€ 71,594,938,845.28
311	***Diversification into non-agricultural activities***	***€ 1,488,899,856.00***	***€ 2,394,086,851.83***
312	Support for business creation and development	€ 2,208,788,801.00	€ 3,093,336,414.58
313	Encouragement of tourism activities	€ 1,291,017,104.00	€ 1,891,732,348.66
321	Basic services for the economy and rural population	€ 3,120,183,405.00	€ 4,557,197,715.83
322	***Village renewal and development***	***€ 3,107,941,407.00***	***€ 4,266,474,624.74***
323	***Conservation and upgrading of the rural heritage***	***€ 1,314,598,779.00***	***€ 2,193,492,811.72***
331	***Training and information***	***€ 147,529,893.00***	***€ 260,780,844.75***
341	Skill-acquisition and animation measure with a view to preparing and implementing a local development strategy	€ 150,021,451.00	€ 258,196,753.24
Total Axis 3		€ 12,828,980,696.00	€ 18,915,298,365.36
411	Competitiveness	€ 471,879,819.00	€ 752,108,267.74
412	***Environment/land management***	***€ 167,031,778.00***	***€ 296,367,242.87***
413	***Quality of life/diversification***	***€ 3,877,472,891.00***	***€ 6,176,672,756.00***
421	***Implementing cooperation projects***	***€ 278,555,888.00***	***€ 460,433,116.26***
431	Running the LAG, skills acquisition, animation	€ 960,569,194.00	€ 1,496,629,140.79
Total Axis 4		€ 5,755,509,570.00	€ 9,182,210,523.65
511	Technical assistance	€ 1,877,371,428.00	€ 2,872,113,979.19
611	Complementary direct payments	€ 645,581,697.00	€ 806,977,121.25
Total	Total	€ 96,275,292,812.00	€ 154,940,370,270.78

Source: European Network for Rural Development

Measures with potential to contribute towards ecosystem restoration are highlighted in ***bold italic***.

8.2.5 Measures under Pillar 1 of the CAP

Beyond rural development there are a number of measures under Pillar 1[120] of the CAP that have the potential to contribute towards ecosystem restoration objectives. These include cross compliance under Articles 5 and 6 and specific support through Article 68. Cross compliance provides a base level of environmental requirements that are a condition of receipt of direct payments. These are set out under Article 5 relating to the Statutory Management Requirements (SMRs), Article 6(1) relating to the maintenance of agricultural land in Good agricultural and Environmental Condition (GAEC) and Article 6(2) on the maintenance of permanent pasture. Article 68[121] allows Member States to provide special support to specific types of farming or activities that are important for the protection of the environment.

Article 5 requires farmers in receipt of direct support payments to abide by national and EU legislation including the Nitrates directive, Birds and Habitats directives which represent part of the environmental baseline on which restoration actions can build. As this measure only seeks to ensure a conditional link with existing legislation and does not propose new measures or practices it will not be considered further in this section.

Article 6 GAEC requirements, similar to the SMR require farmers in receipt of direct support to comply with certain management practices and requirements such as the provision of buffer strips next to water courses or the retention of landscape features[122]. Within the current framework there are eight compulsory standards and seven optional standards that Member States can choose to apply. Those relevant to biodiversity are listed in Annex 2. From 2005, GAEC standards have been mandatory on the whole farm. The conditional GAEC standards such as the retention of landscape features, the protection of permanent pasture, and avoiding the encroachment of unwanted vegetation on agricultural land are all preconditions for restoration activities and help to prevent ecosystem degradation. The optional GAEC standards, where applied, such as minimum stocking rates or the establishment or retention of habitats also have the potential for significant biodiversity benefits. Poláková *et al* (2011, p.85 Box 17) provide some examples of how GAEC standards for minimum level of maintenance have been used to benefit grazed semi-natural habitats.

Both the SMR and GAEC requirements are part of the environmental reference level, and though this is important as a base on which further management can build, they are themselves relatively limited in their potential for ecosystem restoration. This is not to say that simple management operations all have limited potential. For example, ecosystem restoration may in some cases be facilitated by simply withdrawing agricultural management from a blanket bog and as such cost very little to implement. This of course depends on the current status of the ecosystem and the cause of its degradation.

[120] As outlined under Council Regulation 73/2009
[121] Article 68(1)(a)(i) and Article 68(1)(a)(v)
[122] See Annex III of Council Regulation 73/2009

Perhaps the most important measures for biodiversity protection under Pillar 1 is provided through Article 68 funding. Member States are permitted to use up to ten % of their total direct payments envelope, to encourage specific types of farming which are important for the environment, quality production and marketing and/or to buffer the consequences of decoupling of support in particularly sensitive sectors (Poláková *et al*, 2011). In the EU, ten Member States[123] chose to employ the options for special support to farming which is important for the protection of the environment (Article 68 (1)(a)(i)) and for agri-environment benefits (Article 68 (1)(a)(v)), including a range of actions such as extensive grazing, organic farming, maintenance of permanent pasture, crop rotations, and local breeds. There is a relative paucity of information surrounding the use and effect of the Article 68 measure, however some case specific examples do exist and demonstrate the potential effectiveness of such an approach (see Box 8.2).

Box 8.2: Use of Article 68 measures in Ireland

Burren Farming for Conservation

The Burren Farming for Conservation Programme (BFCP) has resulted from the favourable outcomes demonstrated through a LIFE+ funded BurrenLIFE Project (BLP), which aimed to develop a model of sustainable agriculture for the Burren region.

The Burren (from the Irish Boireann meaning 'place of stone') is an area of limestone karst of over 72,000ha, located in the mid-west of Ireland on the Atlantic coast. The area is reliant upon extensive grazing to maintain the mixed karst pasture and prevent scrub encroachment. In recent years, a number of changes have threatened this relationship to the detriment of the environment. Farmers have been increasingly required to take on additional work to supplement farm incomes that have led to less time to access remote areas. At the same time there has been a move away from a mixed farm system based around beef cattle 'stores' to one almost completely dominated by suckler cows that are fed on silage rather grazing the karst.

In 2009 as a result of the favourable outcomes of the BLP the Irish Government announced the BFCP. The BFCP is funded under Pillar 1 of the CAP by the Department of Agriculture with a budget of €1 million per year over four years (2010-2013) using funds under Article 68(1)(a)(i) of EU Regulation 73/2009. Its objectives include ensuring the sustainable agricultural management of high nature value farmland across the Burren and maintaining or enhancing the conservation status of Annex I habitats. While participants are provided with advice on how to maximise the environmental benefit from their land (via a site visit, development of farm plans and provision of best practice guidance), farmers are expected to use their own initiative to create the optimal crop of species-rich grasslands.

The scheme is structured around three measures for which farmers can receive compensation. These measures are:

- Production of species-rich limestone grassland;
- Capital enhancement works (including scrub removal) on Annex I habitats;
- Protection of designated land and other areas of Annex I habitat;
- Requirements of payments include the cessation of silage feeding in all Annex I habitats (both those designated and not designated) and meeting cross compliance and GAEC requirements on the whole farm. Payments are made only following satisfactory compliance checks of outcomes delivered.

[123] Denmark, Finland, France, Ireland, Italy, the Netherlands, Poland, Portugal, Spain, Romania, and the UK-Scotland

8.2.6 Financing measures under Pillar 1 of the CAP

In general, Pillar 1 direct payments are not paid for specific environmental or rural development practices and therefore linking expenditure to specific measures is problematic. Even with Article 68 payments not all Member States utilise this option and the exact proportions of the expenditure vary. Any cost estimates for the use of Article 68 funding would need to be made on a case-by-case basis.

8.2.7 Opportunities and risks under the future CAP

On 12 October 2011 the European Commission published the draft legislative proposals for the future of the CAP post 2013, including significant changes to both Pillars. The proposals have been debated in the Commission, Council and the European Parliament and further evolution of the draft legislative texts has taken place. Furthermore the multi-annual financial framework (MFF), and thus the overall CAP budget, has only recently been agreed. For the CAP, the trilogue negotiations were officially concluded on 26 June 2013, however there remained some outstanding points. Three elements of the reform package remain outside of the scope of this political agreement. These are: transfers between funds, degressivity (the revised approach to capping), and financial discipline. Although political agreement has also been reached for the MFF for 2014-2020, the details, which would provide insight into these three elements, are yet to be disclosed (Hart and Menadue, 2013) Therefore it is necessary to confine our commentary here to the information set out in the proposals released in October and the agreed outcomes of the trilogue negotiations where these are known. However, the final consolidated versions of the legislative texts have not yet produced and so some of the details are not yet completely clear. Therefore what is below is based on our interpretation of the various texts we have seen to date, as they relate to target 2.

There are several significant changes proposed by the Commission that may affect restoration targets. They include the incorporation of environmental objectives within Pillar 1, implemented through greening measures for direct payments[124], and the restructuring of the rural development regulation[125] around six priorities as opposed to the four axes of the current regulation. Monitoring provisions and cross-cutting measures, such as the Farm Advisory System (FAS) and Monitoring and Evaluation System (the successor to the CMEF), can play an important role in helping to meet restoration priorities. These instruments, will also be further refined.

This commentary has been structured around four key issues that have been part of the current debate and that are critical for restoration targets. These are:

- the CAP budget allocation;
- the restructuring of the rural development regulation;
- the distribution of rural development funds and the balance of objectives; and

[124] COM(2011)625/3
[125] COM(2011)627/3

- the greening of Pillar 1 direct payments.

The current budget negotiations

One of the biggest uncertainties surrounding the CAP negotiations has come from the parallel negotiations on the MFF for the next period. The debate of the MFF held during a special European Council meeting (23 November 2012) continued to support the view that the *'total level of expenditure proposed by the Commission, including all elements inside and outside of the MFF, will have to be adjusted downwards.'* This includes the CAP expenditure (under Heading 2 of the MFF negotiating box). For the CAP, it was expected that '*the contribution to the overall reduction has to come from both Pillars'*[126]. It has since been realised, however, that these have not been distributed between the two pillars equally and the proportionate decrease in each pillar has remained the same as proposed (a greater cut being taken from Pillar 2 rural development budget compared to Pillar 1)[127]. In addition, the next MFF contains no discretionary reduction in CAP Pillar 1 expenditure, over and above what a continuation of current rules would imply. To put this into context, the discretionary reduction in Pillar 2 allocations would be around 18% (after Matthews, 2013). This has significant implications for those rural development measures with the greatest potential to support restoration priorities, such as agri-environment climate and forest-environment climate measures. When the proposals were first released, it was understood that the MFF would see the rural development budget suffer a decline in real terms for 2014 – 2020. This could have been countered by those countries that take advantage of the option to move 5% of Pillar 1 funds to Pillar 2 (Baldock and Hart, 2011). However, the draft proposal (and agreed MFF) provides the opportunity for a selection of Member States to be able to transfer funds from Pillar 2 to top up their Pillar 1 funding where the level of direct payments remains below 90% of the EU average. This is of significant concern. Implementation of this transfer, and the disproportional cut to the Pillar 2 budget, would see an inevitable reduction in the overall level of rural development funding available to support restoration measures. The full and final detail of the MFF negotiations is yet to be realised.

The restructuring of the rural development regulation

The draft agreement for the future EAFRD[128] includes several notable changes in relation to rural development. One of the most significant is the replacement of the four axes, which are characteristic of current rural development policy, with six EU priorities[129] relating to the thematic objectives of the Common Strategic Framework (CSF) (Allen *et al,* 2012b). These priorities help to focus rural development funding and form the basis of rural development programming, including the definition of target indicators. With clear relevance to target 2, 'restoring, preserving and enhancing ecosystems dependant on agriculture and forestry'

[126] Presidency issues paper: Multiannual Financial Framework 2014-2020 Informal Meeting of Ministers and Secretaries of State for European Affairs Nicosia, 30 August 2012.
http://www.cy2012.eu/index.php/en/file/t4OXQzwQKPz2nxXo9+AUZw

[127] The precise change in allocation depends on the different counterfactual considered.

[128] (COM(2011)627/3)

[129] See Article 5 of COM(2011)627/3

reflects the inclusion of the EU biodiversity strategy[130] amongst the key EU targets and objectives to be addressed by EAFRD and the other CSF funds[131]. However, this is only one of the six targets covered[132] and will have to be established with sufficient weight in each RDP to ensure sufficient resources are devoted to it.

Several notable changes in the rural development regulation could improve the potential for Member States and regions to deliver land management interventions at the scale required to help in the recovery and restoration of certain ecosystems. These include recognition of the benefits of collaborative action at the landscape scale, with higher transaction costs permitted within the payment calculation for group contracts involving more than one land manager. The removal of the four axes presents significant opportunities to move beyond the current framework to a more flexible approach allowing the more integrated use of measures to deliver restoration priorities. If all the proposed changes are to benefit target 2 and to deliver biodiversity as a strategic priority in all regions of the EU-27, then biodiversity will need to feature clearly as one of the key priorities within any revised EAFRD. The wording of these priorities, as with every other element of the proposals, is still to be seen in consolidated texts.

The distribution of rural development funds and the balance of objectives

The agreed rural development regulation has increased the requirement on Member States to devote a proportion of EAFRD funds towards measures delivering environmental and climate benefits. These have increased from 25% in the Commission's original proposals (and in the current programme) to 30% in the agreed text. Although still not legally binding this represents a positive step to ensure that limited funds are not diverted wholly into measures for competitiveness and risk management and that environmental priorities are taken into account (Hart and Little, 2012). However, the list of measures to which the percentage applies is much extended (beyond the scope of the old Axis 2 measures) and effort will need to be made in order to ensure these funds are spent appropriately if they are to deliver restoration targets.

There are further issues for concern, despite this earmarking of funds. For example, there remains significant room for interpretation as to the types of actions that could be supported. For example it is conceivable within the scope of the proposal that support for anaerobic digesters or wastewater treatment facilities could be attributed as benefiting climate or environmental priorities. The use of EAFRD funds in this way could reduce the overall budget available for agri-environment and forest environment measures that are crucial to meeting restoration priorities. Care will need to be taken to ensure that these funds are used in an environmentally coherent manner and that restoration priorities form part of a balanced suite of wider environmental priorities to be addressed by rural development measures.

---

[130] COM(2011)244 final

[131] COM(2011) 615 final/2 Elements for a Common Strategic Framework 2014 to 2020: the European Regional Development Fund (ERDF), the European Social Fund (ESF), the Cohesion Fund, the European Agricultural Fund for Rural Development (EAFRD) and the European Maritime and Fisheries Fund; and SEC(2011) 1141 final; and SEC(2011) 1142 final. Also COM(2011) 615 final/2.

[132] The other five targets cover competitiveness, innovation, risk management, resource efficiency and economic development.

In terms of the potential contribution towards target 2, rural development measures will continue to have the greatest potential. How the measures are interpreted, the overall design of RDPs, the funding allocations, the way in which the measures are implemented and how they are received by land managers will again determine their real potential. Supported through other measures, such as monitoring and advice, it is likely that the agri-environment-climate and the forest-environmental and climate services and forest conservation measures will remain the two most important measures to aid in the restoration of ecosystems (see for example Allen *et al*, 2012b (Allen et al, 2012b)). It will be important to ensure that sufficient funding and support is afforded to these measures to help them fulfil this potential. For the overall impact of proposed rural development policy much depends on the final legal texts.

The greening of Pillar 1 direct payments

One of the most significant elements of the proposed Pillar 1 reforms related to target 2 of the biodiversity strategy is the greening of direct payments. The trilogue negotiations have maintained the Commission's original proposal that Member States are required to allocate 30% of Pillar 1 national ceilings[133,134] as a green payment. These payments would be conditional on three 'agricultural practices beneficial to climate and the environment', crop diversification, maintenance of permanent grassland and Ecological Focus Areas (EFA). Since the Commission's proposals were published 'greening' has been a major area of contention in the on-going CAP reform negotiations. As a consequence the content of the three measures has been comprehensively watered down, making their likely environmental impact minimal (Hart and Menadue, 2013).

Of the three greening measures, crop diversification is expected to contribute relatively few benefits towards biodiversity or restoration targets. The protection of permanent grassland has the potential to contribute some benefits towards the target by helping to maintain the existing areas of high nature value (HNV) grasslands. However, the extent of these benefits depends on the definition of permanent grassland, the management practices permitted, whether or not the requirement is maintained at the farm level (as proposed by the Commission) or operated at the regional or national level. The date at which the ratio of permanent grassland to other land is determined will be 2015. Not all changes here were negative. One important, and last minute addition to this measures stipulates that Member States must designate permanent grassland, including peat and wetlands in Natura 2000 areas and may identify 'further sensitive areas' situated outside these areas where no conversion or ploughing of these areas must take place. This could have important implications for meeting target 2.

[133] As set out in Article 33

[134] It is important to put the financial support for the greening measures in context. For the 2010 financial year the EAFRD support to the agri-environment measure in Pillar 2 equated to approximately €3 billion[134], whereas in the same year greening payments would have equated to approximately €13 billion[134], over four times the amount (Allen *et al*, 2012a). The figures provided here are illustrative and do not provide an accurate indication of the financial contribution towards environmental objectives let alone restoration priorities.

Of the three proposed measures, Ecological Focus Areas (EFAs) if designed and implemented correctly, still has the greatest potential for contributing towards target 2[135]. However with the weakening of EFA requirements, including an expansion of the types of areas that might be considered EFA (including short rotation coppice and nitrogen fixing crops) there are concerns over the way in which this measure might be implemented and the benefits that will be realised. In addition to expanding the potential types of land that might qualify, the proposed 7% of eligible hectares has been reduced to only 5% and only on arable land over 15 ha. Member States may choose to operate up to 50% of the EFA requirement regionally (pooling the EFA area) which could be of benefit in some areas or a risk in others. Significant exemptions are also in place which include: where over 75% of the eligible agricultural area is permanent grassland, used for the cultivation of grasses or other herbaceous forage or with crops under water; and where over 75% of the arable area is used for the production of grass, herbaceous forage, land laying fallow or cultivated with leguminous crops (Hart and Menadue, 2013). Despite this watering down the EFA measure still has some potential for Target 2 (see (Allen et al, 2012a) and (Westhoek et al, 2012)[136]), even if it will not be required on a significant number of farms in the EU.

In conclusion, it seems reasonably certain that the potential impacts of greening measures on the achievement of Target 2 will be more limited under the agreed texts than set out in the Commission's original proposal.

It should be noted that the greening measures presented here focus primarily on existing agricultural land and the environmental features contained within the eligible area in receipt of direct payments. As such there is limited scope for the maintenance and restoration of important semi-natural habitats and ecosystems outside of the farmed area, which could be more adequately delivered through targeted conservation management such as Pillar 2 agri-environment-climate or forest-environment-climate schemes.

Post-2013 Cross compliance requirements

The proposed cross-compliance requirements for the 2014-2020 period, as outlined in Table 8-2 below, retain some important measures that can help maintain ecosystems, including GAEC 8 on the retention of landscape features and new requirements to limit erosion under GAEC 5. However, the final agreed texts have seen the proposed GAEC 7 (protecting carbon-rich soils) removed from cross compliance, as well as proposed Statutory Management Requirement (SMRs) relating to the Water Framework Directive and the Sustainable Use of Pesticides Directive.

[135] The contribution towards biodiversity objectives is cited in Allen *et al* 2012a; Mathews 2012b and European Commission 2011.

[136] The extent of the benefits that can be realised will depend on the management practices permitted as well as the environmental features that can be considered EFA, their location, whether rotational or not, their geographical connectivity, the link with Pillar 2 measures and the provision of advice

Table 8-2: The European Commission's original proposed cross-compliance requirements for environment, climate change, good agricultural condition from 2014

Source: COM(2011) 628/3 Annex II.

Area	Main Issue	Requirements and Standards		
Environment, climate change, good agricultural condition of land	Water	SMR 1	Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (OJ L 375, 31.12.1991, p.1)	Articles 4 and 5
		GAEC 1	Establishment of buffer strips along water courses [a]	
		GAEC 2	Where use of water for irrigation is subject to authorisation, compliance with authorisation procedures	
		GAEC 3	Protection of ground water against pollution: prohibitation of direct discharge into ground water and measures to prevent indirect pollution of groundwater through discharge on the ground and percolation through the soil of dangerous substances, as listed in the Annex to the Directive 80/68/EEC	
	Soil and carbon stock	GAEC 4	Minimum soil cover	
		GAEC 5	Minimum land management reflecting site specific conditions to limit erosion	
		GAEC 6	Maintenance of soil organic matter level including ban on burning arable stubble	
		GAEC 7	Protection of wetlands and carbon rich soils including a ban of first ploughing [b]	
	Biodiversity	SMR 2	Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (OJ L 20, 26.1.2010, p.7)	Article 3(1), Article 3(2)(b), Article 4 (1), (2) and (4)
		SMR 3	Council Directive 92/43/EEC of 21 May 1992 on the conservation of wild habitats and of wild flora and fauna (OJ L 206, 22.7.1992, p.7)	Article 6 (1) and (2)
	Landscape, minimum level of maintenanc e	GAEC 8	Retention of landscape features, including where appropriate, hedges, ponds, ditches, trees in line, in group or isolated, field margins and terraces, and including a ban on cutting hedges and trees during bird breeding and rearing season and possible measures for avoiding invasive species and pests	

[a] The GAEC buffer strips must respect, both within and outside vulnerable zones designated pursuant to Article 3(2) of Directive 91/676/EEC, at least the requirements relating to the conditions for land application of fertiliser near water courses, referred to in point A 4 of Annex II to Directive 91/676/EEC to be applied in accordance with the action programmes of Member States established under Article 5(4) of Directive 91/676/EEC.

[b] Ploughing of wetland and carbon rich land which has been defined in 2011 at the latest as arable land in accordance with Article 2 point (a) of regulation (EC) No 1120/2009 and which complies with the definition or arable land as laid down in Article 4 point (f) of the Regulation (EU) No DP/xxx shall not be considered as first ploughing.

8.3 Fisheries and Aquaculture Funds

8.3.1 European Fisheries Fund (EFF)

The European Fisheries Fund (EFF) is the primary financial instrument for the provision of public financial aid to the European fisheries sector. It is the key implementing instrument of the Common Fisheries Policy (CFP) (Regulation (EC) No 2371/2002) and it aims to contribute towards attaining the objectives of the CFP, that is, to 'ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions'. Established by Regulation (EC) No 1198/2006, the EFF spans the current programming period (2007-2013), and aims, among other economic and social objectives, to 'promote a sustainable balance between resources and the fishing capacity of the community fleet', and 'foster the protection and the enhancement of the environment and natural resources where related to the fisheries sector'. It is comprised of four priority axes: 1) measures to adapt the fishing fleet; 2) measures relating to aquaculture and inland fisheries; 3) measures of common interest, including 'collective action' and the protection of flora and fauna; and 4) actions to aid the sustainable development of fishing areas.

In terms of content, the measures defined in the Regulation for which Member States are eligible include restructuring support for the fishing industry; improved processing and marketing of fisheries products; investments in aquaculture related to animal health, product quality and environmental improvements; measures to protect and develop aquatic fauna and flora; and the sustainable development and improvement of the quality of life in coastal areas. These measures are listed in Table 8-3, accompanied by the funds committed to them. More specifically, the EFF contains a number of measures which contribute towards the Target 2 objective of maintaining and restoring ecosystems and their services (see those measures highlighted in green in Table 8-3). These include tailored interventions that seek to restore and maintain marine ecosystems and their services, and more commonly, measures which help to reduce the pressures on ecosystems that result from fishing and aquaculture practices.

The greatest pressure to ecosystem services generated by fish populations is overfishing, thus all measures within the EFF which aim to contribute towards balancing fishing capacity with available resources may help to achieve the Target 2 objective. The main measures to this end are scrapping of fishing vessels, and reassignment of fishing vessels, within the framework of fishing effort adjustment plans. Reassignment of vessels can either be to activities outside fishing, or for the purpose of creating artificial reefs, which can have an added environmental benefit in addition to (in theory[137]) reducing the size of the Community fleet (Art. 23).

[137] Although these measures are in theory directly relevant to Target 2, in practice their implementation has proven to be ineffective at reducing the size of the fleet (European Court of Auditors, 2011), and in practice the subsidy has generated perverse investment incentives. There is evidence to suggest that only a small fraction of the subsidies paid out have been used to take active fishing vessels out of service. Loopholes have allowed fishermen to spend the rest on upgrading their ships or on retiring vessels that were either old or already out of service (European Court of Auditors, 2011).

There are also a number of EFF interventions which relate to improving the selectivity of fishing gear. Either through replacement or modernisation these measures aim to improve selection of target species, reduce the impact of fishing on non-commercial species, or reduce the impact of the gear on marine ecosystems, particularly the seabed (see Art. 25). Technical innovations, including improvements in gear selectivity are also available for small scale coastal fleets (see Art. 26) and Member States may apply for funding to support pilot projects investigating novel gear innovations for reducing bycatch, discarding and damage to the sea floor (Art. 41). In addition there is support available for collective actions which aim to remove lost gear from the marine environment so as to combat the phenomenon of ghost fishing.

Other measures within the EFF are also intended to protect aquatic flora and fauna and enhance the marine environment. Such measures range from the construction of static or moveable facilities to protect flora and fauna; the rehabilitation of inland waters such as spawning grounds and migration routes; and the protection and enhancement of the marine environment in the framework of Natura 2000 (Art. 38). In a similar vein, public aid may be available to support pilot projects to enable tests to be carried out on management plans, including the establishment of no-fishing zones, or experimental restocking (Art. 41).

There are also ecosystem maintenance and restoration measures provided for under the EFF aimed at the aquaculture sector. Fish farmers are eligible for compensation if they employ methods which protect and enhance the environment, natural resources, genetic diversity, and the landscape of aquaculture areas, including compensation for aquaculture production methods with specific environmental constraints arising from the designation of Natura 2000 areas (Art. 30). Furthermore, the EFF may support investments in the construction, extension and modernisation of aquaculture facilities if they contribute to substantially reducing negative impacts of aquaculture on the environment, or enhancing positive effects on the environment compared with normal practice in the sector (Art. 29).

Table 8-3 Funds committed by 31 October 2010 broken down by measure

*Key: Measures marked ** and shaded in dark green are entirely relevant to Target 2; measures marked * and highlighted in paler green are partly related to Target 2 but include other social or economic sub measures too.*

Axis	Measure	Commitments 31 Oct 2010 (€) EFF	% of total EFF
1	1.1: Aid to permanent cessation of fishing activities**	263 102 164	22%
1	1.2: Aid to temporary cessation of fishing activities**	137 502 512	11%
1	1.3: Investments on board and selectivity**	44 061 076	4%
1	1.4: Coastal fisheries*	2 885 809	0%
1	1.5: Socio-economic compensations	19 093 613	2%
2	2.1: Investments in aquaculture*	136 713 089	11%

2	2.1.1: Aqua-environmental measures**	32 282 008	3%
2	2.1.2: Animal health measures	0	0%
2	2.2; Inland fisheries	4 436 830	0%
2	2.3: Fish processing and marketing	213 113 246	18%
3	3.1: Common measures	88 068 095	7%
3	3.2: Protection of aquatic flora and fauna**	38 220 203	3%
3	3.3: Port infrastructure	147 170 530	12%
3	3.4: New markets and promotion campaigns	44 564 331	4%
3	3.5: Pilot projects**	41 697 842	3%
3	3.6: Modification for reassignment of fishing vessels	477 013	0%
4	4: All	0	0%
	TOTAL Axis 1, 2 & 3	**1 213 388 362**	**100%**

Source: Adapted from (Ernst & Young, 2011).

8.3.2 European Maritime and Fisheries Fund

As stated above the EFF will run until the end of 2013, when it will be replaced by the European Maritime and Fisheries Fund (EMFF). The EMFF will cover the programming period ending in 2020, but is currently in its formative stages with a proposed draft regulation under discussion within the Council and European Parliament (COM (2011)804). The EMFF will replace the EFF and a number of other financial instruments, including those supporting the Integrated Maritime Policy (IMP). The proposed EMFF budget amounts to €6.5 billion for the seven year period, to which €916 million will be added to finance external fisheries agreements and the compulsory contributions to regional fisheries management organisations. Of the €6.5 billion, €1 billion will be earmarked for the IMP, which leaves fisheries with a marginally greater budget at fixed value than what is available under the EFF (€4.3 billion). The objectives of the proposed regulation are to promote sustainable and competitive fisheries and aquaculture; foster the development and implementation of the IMP, in a complementary manner to cohesion policy and to the CFP; promote balanced and inclusive territorial development of fisheries areas (including aquaculture and inland fisheries); and to contribute to the implementation of the CFP. Unlike the EFF the proposed EMFF is structured into four pillars: 1) smart, green fisheries; 2) smart, green aquaculture; 3) sustainable and inclusive territorial development; and 4) IMP (to support marine knowledge, marine spatial planning, adaptation to climate change, etc).

Bearing in mind the regulation is in draft form, the EMFF, in contrast to the EFF, aims to place a greater impetus on the IMP, to simplify programming and therefore improve access to funds, and coordinate with other structural funds more closely. It puts greater emphasis on the social dimension, and links more strongly to the objectives of the CFP and other pieces of environmental legislation through greater conditionality of funds. More specific changes in measures, relevant to the Target 2 objective of maintaining and restoring ecosystems and their services, are discussed below. Unfortunately, unlike the previous section on the EFF, a detailed breakdown of funding (to the measure level) is not possible at this early stage.

In terms of balancing fishing capacity with available resources, the proposed EMFF no longer provides support for the scrapping of vessels, as it was shown to be ineffective and even counterproductive (see (European Court of Auditors, 2011)). Instead the proposal provides support to systems of transferable fishing concessions (TFCs)[138], established under the CFP. TFCs are proposed in the draft CFP regulation (COM(2011)425) as mandatory for all Member States for vessels over 12m in length, and vessels under 12m in length fishing with towed gear, as the primary means of reducing the overcapacity of the EU fleet. In accordance therefore with the CFP, the proposed EMFF provides support to public authorities for the administrative and technical means of developing the systems, as well as stakeholder participation in this process, and the management, monitoring and evaluation of the systems when in place (Art. 34). These proposals have been objected to widely by Member States, MEPs and even NGOs, although the opposition has not been universal. At a recent meeting of the Council of Ministers it was agreed, as argued by many but not all delegations, that TFCs should be voluntary, not mandatory as stated in the proposal. Furthermore, some Member States and MEPs continue to argue for retaining support for the scrapping of vessels, despite the evidence showing its failure to redress overcapacity.

With respect to reducing unwanted catches, the EMFF proposal supports investments in equipment that will improve the size or species selectivity of fishing gears, and limit the physical and biological impacts of fishing on the ecosystem or the seabed (Art. 36 and 37). This is very similar to the support provided by the EFF; however the EMFF also provides funds for investments on aboard vessels to make best use of unwanted catches of commercial stocks. These new measures are to support the implementation of a discard ban, proposed in the draft CFP regulation. The future of these measures depends very much on whether a discard ban will be adopted within the CFP regulation. Although still under negotiations, the Fisheries Council have agreed on a provisional general approach to the CFP in which a gradual, case by case ban was included. Public support for a ban is high and it is likely to go through – however the success of the discard ban on reducing bycatch will rely heavily on the details of implementation.

Other measures relevant to Target 2 are those intended to protect aquatic flora and fauna and enhance the marine environment in both the fisheries and aquaculture contexts. In terms of sustainable fishing, the proposed EMFF supports measures very similar to those supported by the EFF, but with more emphasis on the participation of fishermen in the protection and restoration of marine biodiversity (see Art. 38). With respect to sustainable aquaculture, the proposed EMFF is more substantial than the EFF, and the funding allocations are likely to reflect this. The proposal includes support for aquaculture with a high level of environmental protection, in order to reduce the impact of aquaculture enterprises on water quality and on nature and biodiversity, reduce water inputs, increase energy efficiency, and restore existing ponds through silt removal or prevention (Art. 52). There is also an article (54) dedicated to

[138] Transferable fishing concessions are a 'cap and trade' mechanism of regulating fishing, where the regulator sets a species-specific total allowable catch, or limit on effort, and a share of that right to fish (ie a concession) is allocated to individuals, vessels or companies. Shares can typically be bought, sold or leased.

fostering environmental service provision through aquaculture. This would provide funds to support additional requirements on farmers resulting from the designation of Natura 2000 areas (as did the EFF), but also provide support for extensive aquaculture, including conservation and improvement of biodiversity, and participation in ex-situ conservation and reproduction of aquatic animals within the framework of conservation and biodiversity restoration programmes. The proposed EMFF also supports conversion to eco-management and audit schemes, and organic aquaculture (Art. 53), as did the EFF. These proposals are not contentious and appear to be supported by Member States and other stakeholders as they have not attracted much attention in the on-going discussions.

8.4 Structural Funds and the Cohesion Fund

8.4.1 Introduction

The implementation of the EU Cohesion Policy is supported by the Structural Funds (ie European Regional Development Fund – ERDF and the European Social Fund – ESF) and the Cohesion Fund. A number of opportunities are, in principle, available under ERDF and ESF for supporting the implementation of the Target 2 of the EU Biodiversity Strategy. The Cohesion Fund (CF) primarily supports large infrastructure projects and therefore it is generally considered as a very unlikely funding source for biodiversity. However, there might be situations where the goals related to biodiversity conservation indirectly profit from projects funded by the CF.

It is important to note that the eligibility for funding under Structural Funds and the Cohesion Fund is linked with supporting broader sustainable socio-economic development and territorial cohesion within the EU. ERDF is generally aimed at strengthening competitiveness and innovation, creating jobs and promoting environmentally sound growth whereas and ESF focuses on promoting social inclusion, education and training, and building institutional capacity (eg creating novel employment opportunities). Therefore, even though ERDF and ESF can be accessed by a wide range of stakeholders actions supported by these instruments need to be linked with the broader sustainable development of the region. Also, funding is not commonly available for on-going / long-term activities aimed at managing biodiversity, ecosystems or related services.

As with most of the EU funds, ERDF and ESF are divided between Member States through national allocations. Member States then allot funding differently between the various budget categories available, including those for biodiversity under ERDF (see below). No compulsory earmarking of funds to support biodiversity under the Structural Funds exists. Also, the budget categories available for – or directly relevant to - biodiversity are very broad leaving a significant room for Member States to decide what kind of measures funding will be targeted. Consequently, concrete possibilities and amount of total funding available for biodiversity, including Target 2 of the EU Biodiversity strategy, will be decided at the national level, depending on national priorities, interest and political will.

8.4.2 ERDF measures with the potential to benefit biodiversity

In terms of opportunities made available for biodiversity in the context of the current EU-level framework (ie Regulation for ERDF and ESF), a general reference is made under ERDF for investment to promote biodiversity and nature protection within different areas of the EU (eg Articles 4(4) and 5(2)b). Also, a number of priority areas with inexplicit relevance to biodiversity and ecosystem services are listed, including prevention of environmental risks and promotion of natural heritage and assets to support tourism (Articles 4(4), 4(6), 6(2)b). In this context direct references are provided to investments in enhanced water supply, water and waste management, wastewater treatment, air quality, prevention of desertification, and aid to mitigate the effects of climate change.

In particular, support is foreseen to be provided for transnational cooperation to with the aim of protecting and managing river basins, coastal zones, marine resources, water services and wetlands (Article 6(2)b). Similarly, financial support to cooperation between Member States is encouraged to mitigate risks of fire and drought, improve flood prevention and enhance natural heritage in support of socio-economic development and sustainable tourism. If taken up appropriately (ie utilising synergies with biodiversity conservation) these EU-level priorities provide a number of possibilities for funding actions that aim to maintain or restore wider ecosystems and their capacity to maintain ecosystem services.

A number of concrete, pioneering projects already exist that use the ERDF funding for the maintenance and restoration of ecosystems and their service, in particular in the context of the ERDF Territorial Cohesion (Art 6 above, also so called 'Interreg' projects). Examples of such projects are provided in Box 8.3 below.

Box 8.3. Examples of the use of ERDF to support the maintenance and restoration of ecosystems and related services Source: (Hjerp et al, 2011)

The emphasis of the ERDF Interreg **Natureship** project (Finland, Sweden, Estonia and Latvia) of Central Baltic Interreg IVA Programme is a novel approach on planning and management of traditional rural landscapes and selected coastlines. The aim of the project is to create and restore an optimal ecosystem service network based on integrated sustainable coastal planning. The project will also assess how to achieve cost-effective planning and management of traditional rural biotopes of city areas in order to enhance public and biodiversity values.

TIDE (Tidal River Development) is an ERDF Interreg project which covers the estuaries of the Rivers Elbe (Germany), Humber (England), Scheldt (Belgium and the Netherlands) and Weser (Germany) and brings together experts, scientists, policy-makers and managers representing economic, social and environmental interests in the four estuaries. The aims of TIDE are to identify knowledge gaps in hydrology, morphology and ecology, integrate planning in local policy whilst ensuring that Natura 2000 and Water Framework Directive requirements are met, and define the most important ecosystem services in each estuary and then relating this to benefits. A budget of €3.7 million is available, 50% of which is derived from the European Regional Development Fund, financed through the Interreg IV B North Sea Programme, and 50% is paid by the partners.

As for the European Social Fund (ESF), it has been acknowledged that some of the activities supported can have a positive effects on environmental management, for example, by providing support for capacity building aimed at the creation of new job opportunities related to the restoration of ecosystems (eg implementation of Natura 2000) (Kettunen et al, 2011). However, the EU ESF framework does not provide any dedicated objectives directly relevant for biodiversity or Target 2 of the Biodiversity Strategy.

Given the above, it appears that ERDF has a significant potential for supporting the implementation of Target 2 of the EU Biodiversity Strategy. However, in order for these opportunities to materialise they need to be taken up at the national level (see below). In addition given the general requirements for eligibility (above), the EU Structural Funds are likely to be most suitable for funding one-off activities related to the maintenance, improvement or restoration of ecosystem's ability to support broader socio-economic benefits, ie different ecosystem services (eg purification of water, mitigation of flooding and enhancing recreational opportunities via development of infrastructure).

8.4.3 Amount of ERDF funding made available for biodiversity in 2007-2013

In terms of concrete budget allocations, the category most directly relevant for biodiversity under the Structural Funds is the category for 'promotion of biodiversity and nature protection (including Natura 2000)', acting predominantly through ERDF (Category 51) (Kettunen et al, 2011). In addition, the categories for promotion of natural assets (Category 55) and protection and development of the natural heritage (Category 56) are relevant for biodiversity and they can be used to maintain and restore a range of ecosystem services, such as ecosystem's ability to regulate water quality and provide recreational, aesthetic and cultural enjoyment. A number of other funding categories also have potential to confer indirect benefits for biodiversity, as well as risks (including ERDF categories 45, 47, 53, 54).

At the national level, there are clear differences in how Member States use the most biodiversity-specific category (ie category 51) (Kettunen et al, 2011). For example, some Member States favour investment in ecotourism or environmental GIS systems whereas others opt for direct conservation actions like habitat restoration or supporting the management of the Natura 2000 network. Therefore, funding for biodiversity under the Structural Funds appears to be allocated through different channels in different Member States and it is very difficult to estimate / get an overall picture on the exact amount of and priorities for the funding (eg the amount of funding directly supporting the implementation of Target 2 of the EU Biodiversity Strategy).

However, to provide a rough indication of the scale of (~max) funding available, Table 8-4 shows the national allocations to category 51 and the combined allocation to categories 51, 55 and 56, as well as the total funding available under the Structural Funds (Kettunen et al, 2011). It can be seen that the scale of the combined allocation of the three most relevant streams for biodiversity (and Natura 2000) varies considerably between Member States. Broadly speaking, however, the allocations appear to reflect the size and economic strength of individual Member States. Given the purpose of the Structural Funds, this pattern is to be expected. As a proportion of the whole funds allotted, the share of categories 51, 55 and 56 is very small, in the range zero to three %. The proportion of funding devoted to these categories varies by Member State.

In general, the current potential spending for biodiversity under the Structural Funds appears to be less than one % of the total budget for European Cohesion Policy. This indicates that there is significant room to increase the level of biodiversity related funding under these funds, especially for projects with a focus on preserving the natural resource base and maintaining valuable ecosystem services.

Table 8-4 Structural Fund (SF) allocations 2007-2013 promoting nature conservation

Source: (Kettunen et al, 2011)

Member State	Total cohesion & structural fund allocation (m€)	SF allocation under categories 51, 55 & 56 (m€)	Categories 51, 55 and 56 as % of Total allocation	SF allocation under category 51 (m€)	Category 51 as % of Total allocation
Austria	1204.5	1.5	0.1%	0.0	0.0%
Belgium	2063.5	25.2	1.2%	1.1	0.1%
Bulgaria	6673.6	159.1	2.4%	80.8	1.2%
Cyprus	612.4	0.0	0%	0.0	0.0%
Czech Republic	26302.6	737.9	2.8%	605.9	2.3%
Denmark	509.6	12.3	2.4%	0.0	0.0%
Estonia	3403.5	46.2	1.4%	21.7	0.6%
Finland	1596.0	16.9	1.1%	1.9	0.1%
France	13449.2	327.5	2.4%	175.2	1.3%
Germany	25488.6	193.3	0.8%	50.6	0.2%
Greece	20210.3	233.3	1.1%	179.8	0.9%
Hungary	24921.1	402.9	1.6%	125.8	0.5%
Ireland	736.5	3.5	0.5%	3.5	0.5%
Italy	27965.3	392.8	1.4%	57.1	0.2%
Latvia	4530.4	26.0	0.6%	26.0	0.6%
Lithuania	6775.5	188.3	2.8%	71.8	1.1%
Luxembourg	50.5	0.0	0%	0.0	0.0%
Malta	840.1	25.1	3.0%	1.7	0.2%
Netherlands	1660.0	22.9	1.4%	5.7	0.3%
Poland	65221.9	306.4	0.5%	135.1	0.2%
Portugal	21411.6	214.9	1.0%	47.0	0.2%
Romania	19213.0	351.4	1.8%	172.0	0.9%
Slovakia	11360.6	76.8	0.7%	30.5	0.3%
Slovenia	4101.0	97.2	2.4%	49.6	1.2%
Spain	34657.7	813.4	2.3%	681.8	2.0%
Sweden	1626.1	3.8	0.2%	2.0	0.1%
UK	9890.9	89.0	0.9%	0.1	0.0%
Total	336,476.17	4,767.44	0.01 %	2,526.65	0.01%

Notes: Funding for Structural Funds refers to Community funding only. To evaluate MS and total public funding a detailed analysis of the operational programmes would be needed, and not all Member States provide this information broken down by categories (eg nature protection). In addition, funding under the territorial cooperation objective has not been included. Categories refer to the priority areas for funding allocations notably category 51: Protection of biodiversity and nature protection (including Natura 2000); Category 55 - promotion of natural assets; Category 56 - protection and development of natural heritage

Source: BAP 2010 Assessment - Country Profiles; DG Regio unpublished data extracted from official national reports

8.4.4 ERDF funding for biodiversity in 2014-2020

The proposed framework for the Structural and Cohesion funds in the upcoming EU budget 2014-2020, including the draft EU Regulations for individual instruments, was published in 2012 with a view to enter into force on 1 January 2014. This framework sets out the overall scope of and investment priorities for the funding of Cohesion Policy.

At the EU level, the funding under ERDF will be adopted and implemented as a part of the broader Cohesion Policy. While the Common Provisions Regulation (CPR) (COM/2011/615) sets out common provisions for all key EU funds (ERDF, ESF, Cohesion Fund, EAFRD and EMFF), a dedicated ERDF Regulation (COM/2011/614) sets out specific provisions concerning the fund, determining the scope of intervention under ERDF and defining investment priorities for the thematic objectives (Box 8.4). In addition, the Regulation also sets out common indicators for monitoring ERDF support, including indicators for environment related investments. A dedicated indicator for nature and biodiversity is also foreseen to be included on the list (IEEP et al, 2012; Kettunen et al, 2012; Medarova-Bergstrom and Volkery, 2012).

As before, the Member States and regions can focus on allocating funds towards those priorities that they consider the most relevant for their future development. The national and regional framework for ERDF implementation in 2014-2020 consists of Partnership Agreements and Operational Programmes, developed based CSF (IEEP et al, 2012; Kettunen et al, 2012; Medarova-Bergstrom & Volkery, 2012). Partnership Agreements (PAs) - with their binding obligations to Member States - will be agreed between the European Commission and each Member State, translating the elements set out in CSF into the national context. Operational Programmes (OPs) are the key planning tool for ERDF expenditure. They contain, for example, a development strategy for the funding covered by the programme, funding priorities and specific objectives and measures, financial appropriations and indicators for monitoring the implementation of funds. OPs are drawn up by Member States and they consist of priority axes, each axis corresponding to a CPR thematic objective and comprising one or more investment priorities related to the given objective. Consequently, the content of OPs will be the key factor defining the opportunities available for biodiversity, including the implementation of Target 2, in the future.

While the main focus of EU funding for cohesion is on promotion of growth and jobs (ie implementing the Europe 2020 Strategy), the key aspect of sustainable development, including environmental protection, resource efficiency, climate change mitigation and adaptation, disaster resilience, and risk prevention and management, are also promoted. These focal areas still provide a number of (direct and indirect) opportunities for financing biodiversity conservation and Target 2 of the Biodiversity Strategy under ERDF.

Protecting the environment and promoting resource efficiency (thematic objective 6, Box 8.4) is the only thematic objective with direct relevance for the implementation of Target 2 under the future ERDF. It clearly identifies biodiversity as an investment priority, providing for funding to be allocated towards investments such as the protection of biodiversity (eg Natura 2000), the promotion of ecosystem services and Green Infrastructure. In addition, other identified investment priorities under this objective, such as broader environmental protection and resource efficiency can, be implemented through measures that also support Target 2. For example, regional water security can be enhanced by creating and/or restoring wetlands, leading to investment in natural - rather than manmade - water purification systems (Kettunen et al, 2012).

In addition to the direct funding opportunities, the proposed ERDF thematic objectives include a number of areas where synergies with other objectives can be used to create funding opportunities for Target 2. For example, supporting the shift towards a low-carbon economy (thematic objective 4) can provide a basis for the restoration of biodiversity and ecosystem services to mitigate climate change. For example, the conservation and restoration of peatlands and forests can both prevent emissions from degraded habitats and/or improve carbon sequestration, therefore complementing actions to mitigate greenhouse gas (GHG) emissions. In addition, urban green areas can play an important role in stabilising local temperature peaks, reducing an area's overall energy footprint.

Similarly, promoting climate change adaptation, risk prevention and management (thematic objective 5) can be used to support ecosystem-based adaptation measures that build on the restoration of ecosystem services. For example, restoring ecosystems' natural capacity to buffer the impacts of climate change (eg frequent heavy rains and other extreme weather phenomena) can be used as a means to mitigate flooding, droughts and wild fires. Investment in such ecosystem-based adaptation measures is explicitly encouraged under the proposed CPR Common Strategic Framework. When appropriately planned, such adaptation measures can also contribute to the restoration of habitats, further delivering on the EU's biodiversity objectives.

In addition to the above, a number of innovative inter-linkages between restoration and welfare can be used to create opportunities for financing the implementation of Target 2. For example, enhancing the competitiveness of SMEs and promoting employment (thematic objectives 3 and 8) can be used to create opportunities for investing in natural solutions such as restoring ecosystem's ability to maintain water quality, providing a cost-effective solution for SMEs to improve their performance and competitiveness. Similarly, promoting social inclusion and combating poverty (thematic objective 9) can, in principle, be supported by restoring ecosystems and ecosystem services to deliver welfare impacts, for example with a view of improving environmental security and/or quality in the area. Restoration of green areas can also support a range of measures and activities that enhance social inclusion, including providing opportunities for nature-based therapy and care.

Finally, in the context of the 11 thematic objectives specific support from the ERDF is foreseen to be given to cities and urban development. For example, the current ERDF proposal envisages ring-fencing 5 % of funding for integrated sustainable urban development measures and for the setting up of an urban development platform to promote exchanges between cities. This focus on cities and urban areas provides a range of opportunities for the restoration of urban ecosystems and related services (eg urban wetlands, water bodies and green spaces).

Box 8.4. Foreseen investment priorities for ERDF under the 11 thematic objectives in 2014-2020

Note: Thematic objective 6 is directly relevant for biodiversity whereas numerous synergy-based and/or indirect opportunities for financing biodiversity can be established under a range of other objectives. Thematic objectives 8-11 are mainly foreseen to be covered by the European Social Fund (ESF).

Thematic objective 1: Strengthening research, technological development and innovation

Thematic objective 2: Enhancing access to and use and quality of ICT

Thematic objective 3: Enhancing the competitiveness of SMEs

Thematic objective 4: Supporting the shift towards a low-carbon economy in all sectors

Thematic objective 5: Promoting climate change adaptation, risk prevention and management, including investment priorities for supporting dedicated investment for adaptation to climate change and promoting investment to address specific risks, ensuring disaster resilience and developing disaster management systems

Thematic objective 6: Protecting the environment and promoting resource efficiency, including investment priorities for:
a) addressing the significant needs for investment in the waste sector to meet the requirements of the environmental *acquis* (body of EU legislation);
b) addressing the significant needs for investment in the water sector to meet the requirements of the environmental *acquis*;
c) protecting, promoting and developing cultural heritage;
d) protecting biodiversity, soil protection and promoting ecosystem services including Natura 2000 and green infrastructures; and
e) action to improve the urban environment, including regeneration of brownfield sites and reduction of air pollution.

Thematic objective 7: Promoting sustainable transport and removing bottlenecks in key network infrastructures

Thematic objective 8: Promoting employment and supporting labour mobility

Thematic objective 9: Promoting social inclusion and combating poverty

Thematic objective 10: Investing in education, skills and lifelong learning by developing education and training infrastructure

Thematic objective 11: Enhancing institutional capacity and an efficient public administration by strengthening of institutional capacity and the efficiency of public administrations and public services related to implementation of the ERDF

Source: (Kettunen et al, 2012)

As for the amount of financing available, while all regions in Europe will receive funding from ERDF, the amount of support will depend on a region's level of economic development (per capita GDP). In 2014-2020 three distinct regions are foreseen to be identified, including less developed regions (GDP per capita less than 75 % of the average GDP of the EU-27), transition regions (GDP per capita between 75 % and 90 % of the average) and more developed regions (whose GDP per capita is above 90 % of the average). In general, the less developed regions are foreseen to receive most of the funding under CP / ERDF (around 40 % of the total CP funding) with contributions to the other regions being considerably less (10 % and 14 % of the total CP funding, respectively). In addition, a certain amount of financing (3 % of the total CP funding) is foreseen to be allocated to support territorial cooperation within the EU (ie cooperation between different regions). These regional ERDF allocations are important also from the perspective of biodiversity as, based on previous experience eg (Kettunen et al, 2011), the overall amount of ERDF funding available within a given Member State or region is also indicative of the scale of ERDF support made available for biodiversity (Kettunen et al, 2012).

8.5 LIFE Programmes

8.5.1 Introduction

The LIFE+ Programme is Europe's dedicated financial instrument directly supporting environmental and nature conservation projects throughout Europe, as well as in some candidate, acceding and neighbouring countries. Since 1992, LIFE has co-financed some 2,750 projects, contributing approximately 1.35 billion euros to the protection of the environment. It represents a limited but focused funding instrument providing specific support for the development and implementation of Community environmental policy and legislation, in particular it is specifically envisaged to support the Birds and Habitats Directive, the implementation of the 6th Environment Action Programme (6EAP) and resulting thematic strategies. This importance of the LIFE instrument was also raised by Environment Ministers when discussing the necessary actions under the EU Biodiversity Strategy in their Council Meeting of 19th December 2011, as one of the most successful and most cost-effective EU funding instruments, delivering through innovative, bottom-up projects significant measurable outcomes for ecosystems and economic development.

The programming stage for LIFE Programme is currently centrally administered at EU level on a multi-annual basis, with the current phase of the programme, LIFE +, running from 2007-2013. It comprises three components: LIFE+ Nature and Biodiversity, LIFE+ Environment Policy and Governance and LIFE+ Information and Communication. For the current programming phase, at least 78% of the LIFE+ budget is intended at the co-financing of project action grants, of which at least 50% have been earmarked for Nature and Biodiversity projects.

Under the post 2013 MFF, the Commission plans to continue LIFE with a new programme running from 2014 to 2020. The Commission proposes to allocate EUR 3.6 billion[139] in current prices (EUR 3.2 billion in constant prices) over to the new LIFE Programme, which is divided into the sub-programmes for the Environment and for Climate Action.

[139] This allocation might change as a consequence of the negotiations of the post 2013 MFF.

The European Commission presented a number of modifications in order to fulfil LIFE EU added-value potential.[140] Main changes include:

- Clearer definition of priority areas within multi-annual work programmes developed by the Commission and adopted in consultation with the Member States;
- Promotion of a new type of projects - "Integrated projects" – aiming at implementing on a large territorial scale environmental or climate strategies or action plans, and mobilising other EU, national and private funds. These will focus primarily on the nature, water, waste, air, and climate change mitigation and adaptation sectors.

Under the current programme, the Commission makes an annual call for eligible project proponents to apply and decides on the list of projects to be financed. The Commission ensures a proportionate distribution of projects by establishing indicative annual national allocations and a set of eligibility criteria. The new proposed Regulation, however, suggests a shift to a more flexible top-down approach. Work programmes valid for at least two years, will be drawn up by the Commission in consultation with Member States based on the multi-annual strategic programme, including specific but non-exhaustive priorities linked to specific targets and assessed by defined indicators. Under the future proposal, there will be a greater focus on the so called integrated projects to help with larger scale implementation matters.

8.5.2 LIFE contribution to habitat management and maintenance

LIFE is a major instrument for habitat management and maintenance, while it directly supports environmental and nature conservation projects. As was highlighted in the results of the first assessment of the conservation status of species and habitats (Article 17 report), which was published in 2010, the importance of LIFE lies in that it is the sole source of funding for the conservation, restoration and management of certain species and habitats at EU level. This continuous source of targeted financing has radically changed the capacity of many countries and regions to care for and manage Natura 2000 sites. Targeting some 2,200 Natura 2000 sites, or approximately 8% of the network LIFE projects have had a significant impact on the Natura 2000 network, by restoring approximately 320,000 hectares in Natura 2000 sites, which have also led to the designation of hundreds of new sites.

Thus although LIFE only represents merely 0.3% of the total EU budget, it remains a particularly relevant financing instrument to attain Target 2 of the EU 2020 Biodiversity Strategy. LIFE project actions are varied and can encompass the development of management plans and other policy documents, support for the enlargement of the Natura 2000 network, improving knowledge of species and habitats, direct conservation actions, capacity building and awareness raising. As was noted earlier in this study, according to a preliminary search of the LIFE database, between 1992 and 2011 almost 1000 LIFE and LIFE+ projects involved restoration, which corresponds to almost 1/3 of all projects financed in the same period. LIFE

[140] COM(2011) 874 final

has also financed more than **1000 management plans** across the Natura 2000 network. For example, the project, 'Regeneration and preservation of dry grassland in Germany' (LIFE00NAT/D/007058), involved the development of management plans for 14 Natura 2000 sub-sites. Several management models implemented by LIFE projects have also been highlighted as best practice examples for habitats.[141]

Within the LIFE Programme, projects falling under the LIFE+ Nature and Biodiversity component are clearly the most relevant for habitat management and maintenance. To date, the biodiversity component (formerly LIFE Nature) of LIFE has co-financed a total of 1,108 projects, providing some €1 billion, and mobilising a further €1 billion in other public and private contributions[142]. The new proposed regulation shares common objectives with the current programme, adding as one of its specific Biodiversity objectives the importance for the upcoming programme "to contribute to the implementation of the Biodiversity Strategy to 2020". The expected stated achievements by the end of the next programming period in the proposed regulation - related to habitat management and maintenance - for the Biodiversity priority area are:

- 25% of habitats targeted by projects improve conservation status
- 25% of species targeted by projects improve conservation status
- 3% ecosystem services restores
- 15% of the Natura 2000 network adequately managed

Moreover, the new LIFE Integrated Projects (IPs) propose a new type of project that could also prove relevant for habitat conservation by improving the integration of environmental aspects in other EU policies. The IPs aim to improve the implementation of environment and climate policy by focusing on the implementation of plans and strategies on a larger territorial scale (eg regional, multi-regional, national).

They are also intended to coordinate the mobilisation of other EU, national and private funds for environmental and climate objectives, encouraging applicants to develop a strategic approach towards certain environmental and climate challenges by using various funds and programmes.[143] IPs for the Environment will focus on plans and programmes will be used to implement new or existing sectoral programmes including for example regional Natura 2000 networks and river basin management plans.

Finally, it must be noted that there is also great potential for synergies between the various components of LIFE programme, between for example environmental and climate objectives as projects can benefit multiple purposes. For example, climate change projects might contribute to habitat restoration and maintenance while they could for instance be aimed at the maintenance at a Favourable Conservation Status of the main carbon sink habitats, namely

[141] http://life.lifevideos.eu/environment/life/toolkit/comtools/resources/documents/NAT_key_msgs.pdf
[142] *Ibid*
[143] http://ec.europa.eu/clima/policies/finance/budget/life/faq_en.htm

peatlands, forests and wetlands while their conservation represents a significant contribution towards limiting greenhouse gas emissions.

8.5.3 Up-to-date LIFE Nature project budget allocations for each Member State

Under the new LIFE programme there will no longer be indicative national allocations, as is the case under the current LIFE+ programme (based on population and on the percentage of the national territory designated as Natura 2000). The new LIFE programme will first and foremost select projects that demonstrate best practices and approaches and add value at the European level. However, a geographical balance will be sought in the grants for Integrated Projects, while some Member States have a greater responsibility vis-à-vis the EU area.

To date no accurate data are currently available on the project budget allocations on previous projects for each Member State. However, Kettunen et al. (2011) attempted to provide a set of available data on the budget allocations to conservation projects approved and established during the 2000 to 2006 funding period under the 'Nature' component of LIFE III and a second dataset referring to indicative allocations foreseen under LIFE+ for the years 2007 to 2010 (Table 8-5). However, the study acknowledges a number of limitations in the presented data, including the fact that for the 2000-2006 period no data was available for 2001, and the data should be regarded as reflecting budgeted allocations only and provides no guarantee that these budgeted commitments reflect the sums actually paid.

Table 8-5 Budgeted allocations by Member State of funds under the LIFE III Nature programme (2000-2006) and indicative allocations under LIFE+ 'Nature and Biodiversity' (2007-2010)

Source: (Kettunen et al, 2011)

Member State	LIFE III Nature budget allocation 2000 - 2006				Indicative allocation 2007 - 2010	
	Total budget (m€)	Member State share (%)	Number of projects	Average budget per project (m€)	Total LIFE+	LIFE+ Nature and Biodiversity*
Austria	24.6	48.5%	80	0.31	16.1	8.1
Belgium	38.5	56.5%	133	0.29	17.7	8.9
Bulgaria	na	na	na	na	18.5	9.3
Cyprus	1.5	42.3%	19	0.08	9.2	4.6
Czech Republic	1.1	26.7%	3	0.37	17.1	8.5
Denmark	23.7	50.0%	73	0.32	21.1	10.6
Estonia	4.2	34.4%	21	0.20	14.2	7.1
Finland	18.5	48.5%	118	0.16	30.8	15.4
France	26.3	48.2%	270	0.10	75.2	37.6
Germany	45.0	43.7%	264	0.17	100.1	50.0
Greece	16.0	36.5%	170	0.09	29.2	14.6
Hungary	12.7	38.0%	35	0.36	21.5	10.7
Ireland	10.2	30.6%	50	0.20	13.5	6.8
Italy	38.0	48.7%	445	0.09	75.7	37.8
Latvia	11.5	32.7%	26	0.44	11.7	5.8
Lithuania	1.5	50.0%	9	0.17	11.2	5.6
Luxembourg	1.1	52.2%	14	0.08	9.8	4.9
Malta	0.5	44.4%	10	0.05	9.9	4.9
Netherlands	18.7	59.3%	146	0.13	27.6	13.8
Poland	6.5	30.1%	7	0.93	40.7	20.4
Portugal	14.6	39.7%	134	0.11	24.1	12.0
Romania	7.0	37.5%	44	0.16	37.4	18.7
Slovakia	4.8	36.8%	12	0.40	13.1	6.6
Slovenia	6.2	40.4%	18	0.34	18.5	9.3
Spain	60.8	48.2%	447	0.14	92.1	46.0
Sweden	11.2	50.2%	88	0.13	35.3	17.7
UK	32.3	47.8%	182	0.18	68.4	34.2
Total	437.0	47.3%	2818	0.16	859.6	429.8

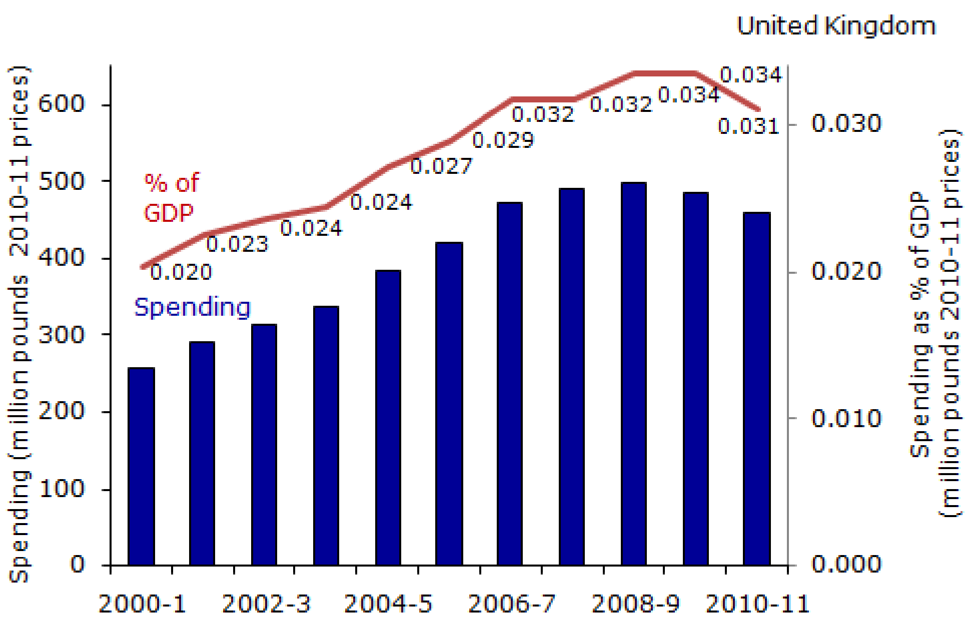
8.6 National public financing

In some EU Member States public funding is provided, for example, through central government (or related agencies) or local authorities for ecosystem restoration projects. Two examples of such funding for large scale projects in the UK and the Netherlands are outlined below.

8.6.1 Public funding for biodiversity restoration in the UK

UK Government and public sector funding is available to meet a large proportion of the costs required for implementing the UK's biodiversity strategy and achieving CBD objectives. This is supplemented by funding from biodiversity-oriented NGOs and the Heritage Lottery Fund[144]. In 2010-11, about £460 million of UK public sector funding was spent on UK biodiversity, which represents a decrease of 5 % compared with 2009-10 (Figure 8-1). This figure includes financing from the EU, National Lottery funding, combatting Wildlife Crime and contributions to biodiversity in national road construction.

Figure 8-1 Public sector expenditure on biodiversity



Source: UK biodiversity indicators in your pocket, 2012[145]

[144] http://uk.chm-cbd.net/default.aspx?page=6873
[145] http://jncc.defra.gov.uk/pdf/BIYP_2012.pdf

Most work on nature conservation was previously carried out under the UK Biodiversity Action Plan (UK BAP). An important component of the UK BAP was the establishment of a network of Local Biodiversity Action Plans in which local Partnerships deliver conservation action and engage with communities. Following devolution in 1998, the four countries of the UK (England, Northern Ireland, Scotland and Wales) developed their own country strategies for biodiversity and the environment, allowing conservation approaches to differ according to the different environments and priorities within the countries.

Detailed analysis of the funding of the BAP was provided in 2006 by GHK consulting, with an update in 2010[146]. The study shows that most significant contributors to BAP related expenditure in each country (with the exception of Northern Ireland) are the agri-environment schemes. The Forestry Commission, Wildlife and Countryside Link organisations, Heritage Lottery Fund, and statutory nature conservation agencies make up the other major contributors within each country.

In England for example, BAP related agri-environmental expenditure in BAPs in 2006 totalled £105 million, covering some 48% of the approximate total of £219 million expenditure on BAPs. This was provided through schemes such as Countryside Stewardship, Environmentally Sensitive Areas, Conservation Plans, Entry Level Stewardship (ELS) and Higher Level Stewardship (HLS). The nature conservation NGO's assembled in the Wildlife and Countryside Link (WCL) are the second largest contributor to BAP expenditure in England. Their direct spending on BAP measures is estimated at £65 to £80m. The majority of this is devoted to maintaining existing populations or areas of species and habitats. The estimated spending by WCL organisations on nature conservation in the whole UK is £195m per year. These budgets are used to manage a substantial part of the UK's existing biodiversity, including hundreds of nature reserves and many UK BAP priority species and habitats. The Forestry Commission (FC) is the next largest contributor. A best estimate suggests that 18.7 million was spent on BAPs, representing 75% of the grants and partnerships budget, and 11% of the Forest Enterprise budget. The Big Lottery Fund and the Heritage Lottery together provided some 13 million, the Environment Agency £9 million and smaller funds were provided by various sources, including EU LIFE.

The UK BAP has now been replaced by the new 'UK Post-2010 Biodiversity Framework' (July 2012), in response to a change in strategic thinking following the publication of the CBD's 'Strategic Plan for Biodiversity 2011–2020' and the launch of the new EU Biodiversity Strategy (EUBS). The framework set out how the work of the four countries and the UK contributes to achieving international; targets, and identifies the activities required to complement the country biodiversity strategies. The UK BAP partnership is no longer in place, although the Local Biodiversity Partnerships and Action Plans are still in operation. The UK biodiversity Framework seeks to further mainstream biodiversity concerns and focus ecosystem services through promoting integrated approaches.

[146] http://archive.defra.gov.uk/evidence/economics/foodfarm/reports/documents/Biodiversity.pdf

Nature Improvement Areas

In the view of climate change and other pressures on England's terrestrial biodiversity, a review was carried out of the adequacy, coherence and resiliency of the network of wildlife sites in England (Lawton et al, 2010). The report 'making space for nature' identified fragmentation as major problem in England, and proposed a new restorative approach to nature conservation which 'rebuilds and creates a more resilient natural environment', thereby establishing a coherent ecological network. Summarised as 'more, bigger, better and joined', the report made 24 wide-ranging recommendations related to better management, increasing the size of wildlife sites, reduce pressures and enhance connections. The report recommended the establishment of large scale Ecological Restoration Zones (ERZs), by enhancing existing wildlife sites, improving ecological connections and restoring ecological processes. These ERZs should be proposed and implemented by consortia of local authorities, local communities and landowners, the private sector and voluntary conservation organisations, supported by national government.

Reflecting this recommendation, the Department of Environment, Food and Rural Affairs provided £7.5 million over 3 years to fund 12 pilot Nature Improvement Areas (NIAs). In its Natural Environment White Paper, the Government acknowledges that people cannot flourish without the benefits and services of the natural environment, and recognises that over 30% of these services are in decline. This justifies the institutional framework that is put in place to achieve the recovery of nature. A national competition was set up to 'see Nature Improvement Areas wherever the opportunities or benefits are greatest, driven by the knowledge and vision of local partnerships' (HM Government, 2011). After several competition rounds, 12 partnerships were selected to receive government funding.

An example of a funded NIA is 'Wild Purbeck', an area with internationally important heathlands, valley wetlands and a World Heritage Site. A partnership of 10 organisations, including NGO's, local government and government agencies (Natural England, Forestry Commission) will endeavour (among other) to increase the natural functioning of the landscape - for example rivers are re-connected with flood plain and catchment-, enable people and wildlife to adapt to sea level rise and increase the connection between the landscape and the local economy while raising awareness and public support. National funding of about £600,000 was provided for this NIA, while £1,490,000 is provided by other sources. The NIA area included a number of Natura 2000 sites such as Dorset Heaths (SPA, SCI) and Studland Cliffs (SCI).

Typically, all 12 selected projects combine restoration activities with increased community participation and awareness, as 'reconnecting people and nature' is a major pillar under the Governments vision for the natural environment. Also, significant amounts of additional funding are provided by the partnerships, to complement the national funding. This way, the relatively modest national investment of £7.5 million has attracted over £40 million of additional funding provided by the bidding partnerships in the competition. However, these figures compares to an estimated annual funding requirement of £600 million to 1 billion that would be needed to ensure ecological resilience across the UK, according to the Lawton Review.

However, it is likely that at least some of the resources allocated to NIAs through the competition would nonetheless have been directed to nature conservation. Traditionally in the UK, a relatively large share of the funding for nature conservation and restoration measures is provided by NGOs, in particular National Trust, the Wildlife Trusts and the RSPB. These organisations are among the biggest of their kind in the world, both in terms of membership and ownership of sites as well as in terms of financial resources. For local governments, it is a legal duty for local governments to have regard to the purpose of conserving biodiversity, as a core element of sustainable development. This also means participating in biodiversity partnerships and developing asset management plans. Consequently, a wide range of nature restoration projects were initiated and funded by partnerships of NGOs and public bodies, sometimes also with private and business contributions, even before the national government initiated the NIA-approach (see case studies below). Although there is no all-inclusive overview of the spending on restoration activities in the UK, the anecdotal case studies below indicate that public-private partnerships play a major role in financing nature restoration in the UK. The new Nature Improvement Areas may make a difference in the sense that the available resources are spend in a more concerted manner. Through the development of partnerships, funding is focussed towards large scale restoration in discrete areas. However, although the competing project proposals were assessed by an independent panel (chaired by sir John Lawton) using objective criteria, such as an assessment of the expected outcomes against the strategic objectives of the NIA, the bottom up approach of the competition does not guarantee that the areas that are most in need of restoration are in fact proposed or selected as NIA.

Conclusion
All in all it seems that the UK's ambitious biodiversity commitments will primarily be reached through country-based strategies, which in turn heavily rely on EU funds and financial contribution from NGOs. As the gap between the investment needs, and the actual expenditure remains significant, one of the priority actions in England's biodiversity strategy for 2020 is to develop new and innovative financing mechanisms to direct more funding towards the achievement of biodiversity outcomes.

8.6.2 National funding of the Dutch National Ecological Network (NEN)

Introduction
The creation of a connected network of nature reserves, the National Ecological Network (NEN), constitutes a major element of the Dutch nature conservation policy. The goal of NEN is to contribute to biodiversity conservation by increasing the area of interconnected nature reserves to promote their resilience and coherence.

The concept of an interconnected network of nature areas was introduced in the Netherlands in 1990 (Nature Policy Plan 1990). It marked a change in nature conservation policy, from a focus on preserving existing nature reserves, to the active restoration and (re)-creation of habitats: ie nature development. Against a background of declining populations and decreasing habitat quality, it was recognised that the remaining nature reserves were too small and too fragmented for the long term survival of habitats and species. Inspired by the island theory (MacArthur and Wilson, 1967), the state of nature reserves in the Netherlands was recognised as isolated islands in a sea of cultivated landscape, vulnerable to pollution and local extinction. With the recognition that the existing policy was insufficient to tackle these problems, a new strategy was introduced to create a coherent network of high quality nature areas by increasing their size, connecting them and improving their quality.

Goal of the NEN

The National Ecological Network was set out to be a long term national project for the conservation of biodiversity by means of the realisation of area of interconnected nature reserves, thereby increasing resilience and promoting the exchange of populations. The NEN was designed to cover 720,000 ha, consisting of 450,000 ha existing nature reserves (including all Natura 2000 sites) and with 270,000 ha of ‘new nature’ to be created through land use change by 2018.

By 2010, some 575,000 ha had been established and brought under conservation management. The ambition was then reduced to a total of 630,000 ha, resulting in a remaining task of 55,000 ha new nature to be created by 2021.

Government expenditure[147]

Government expenditure on the NEN can be divided into three parts: expenditure on acquisition of land, conversion of land into natural habitat and the management of sites. Expenditure on land acquisition was on average 100 million per annum in the period 1990-2009. After an increase from €50-100 million per year to around €200 million a year (due to a more active policy and increase land prices), a larger part of the NEN was to be achieved through agri-environmental agreements (rather than land acquisition), reducing the costs for land acquisition (but increasing the management costs).

Expenditure on the conversion of acquired land for the NEN constitute between €20 to 45 million per year. After 2005 this increased to €130 million, due to a different accounting system. Since 2005 expenditure on improvement of environmental conditions are included in the figure.

Government expenditure on management of NEN sites are on average €129 million per year. Costs increased steadily form €93 million in 1990 to 175 million in 2009, mainly due the increased area. Expenditure on management involves financial support to the maintenance of forest and nature by statutory bodies, NGOs and private organisations. In 2009, some 590,000 ha of the NEN was financially supported (454,000 existing nature reserves, 46,000 newly

[147] Source: compendium for the living environment: http://www.compendiumvoordeleefomgeving.nl/dossiers/nl0067-kosten-natuur-en-landschap.html?i=10-57

developed EHS, and 80,000 ha under agri-environmental agreements). On average the national government contributed 295 €/ha to conservation management.

The total expenditure of the national government on nature and landscape conservation are on average €500 million per year, of which more than half is invested in the NEN. Private nature organisations and business contribute on average €400 million per year to nature conservation. In 2007 NGO activities contributed €137 million to nature conservation, primarily funded by members, legacies and lotteries.

In total, the Dutch national government invested €5.5 billion in the realisation of the NEN between 1990 and 2009. Land acquisition costs accounted for 37%, land conversion 17% and management 46%. This constitutes 0.14% to 0.20 % of the total national government budget of €200 billion a year.

9 POTENTIAL OF PRIVATE SECTOR FINANCE TO CONTRIBUTE TO TARGET 2

This chapter focusses on the potential for innovative or less established funding instruments, including those designed to attract private sector finance, to contribute to reaching Target 2. It has been written with a view to being an accessible overview of the topic and therefore is relatively short compared to preceding chapters. However, considerable additional detail is provided in annex 3 (a detailed description and analysis of the funding instruments) and annex 4 (a selection of case studies on the use of the financing instruments).

9.1 The need for alternative sources of financing for nature conservation

Traditionally, biodiversity conservation has primarily been funded from public funding sources from national and European level (see Chapter 8). However, pressure on public budgets and greater recognition of the economic value of the environment through ecosystem services concepts are leading to greater consideration of the use of private funding to support biodiversity. The mixture of public and private values from biodiversity restoration means that there are few *purely* private sources of funding that can be expected to develop. However, a range of measures exist through which public funding and/or policy actions can potentially stimulate increased private sector funding of biodiversity, thereby supporting policy objectives such as the maintenance and restoration of ecosystems under Target 2.

Biodiversity has historically been considered a liability to business activity. Although perceptions of the value of biodiversity are gradually changing, barriers to private sector investment in ecosystem protection persist, including:

- Low rate of return unattractive to investors who pursue other markets;
- Lack of internationally agreed biodiversity metrics and indicators to measure the positive and negative results of investment; and
- Lack of understanding amongst bank managers of biodiversity-friendly business models and the often insufficient size of the businesses.

Investments by business in biodiversity also require an appropriate framework of laws, regulations, taxes, incentives and social norms to adequately govern the actions of businesses and investors. In addition, investments from rural communities are often omitted from conservation project designs (Gutman, 2003), but can be an important source of investment in rural settings. Communities require good prospects of short-term return with acceptable levels of risk to investments (Dickie et al, 2012).

Some of the more innovative actions (at least in terms of application to the natural environment) were explored in a recent study (Dickie et al, 2012), which examined several potential areas of intervention[148]. This section builds on that analysis, adapting it to the context

[148] The study focused on biodiversity offsets, green infrastructure and bio-carbon but also examined pro-biodiversity businesses, payments for ecosystem services (PES), and cross-cutting issues of risk and information provision.

of this study with the aim of identifying which mechanisms are the most suitable for mobilising private finance to fund actions under Target 2. The section examines how private finance can be attracted for maintaining and restoring degraded ecosystems and establishing green infrastructure; and whether this can be achieved by adapting established financing instruments for biodiversity conservation or by creating new and innovative instruments.

9.2 Overview of potential private sector financing instruments

Many of the mechanisms identified in this study can be placed on a scale depending on the motivation behind private sector actions from purely philanthropic[149] motivation (eg private philanthropic donations) to purely profit motivated (eg payments for ecosystem services). Profit driven motivations relate to actions by businesses that can profit from ecosystem restoration, such as the certification of sustainably-produced food. In this sense 'profit' is defined broadly in terms any type of return such as reputational, financial or reduced risk. Philanthropic motivations, similarly, can range from truly altruistic to reputational management (hence related to profit). Most of the mechanisms currently sit in the middle part of this philanthropic-profit scale, in that while they make some economic sense to the private sector, part of the motivation for them, particularly in pilot efforts, may be altruistic. Regulatory compliance is a third likely motivation for private sector financing.

Different combinations of these will be more or less appropriate for different restoration actions and habitats. Private funding initiatives for biodiversity conservation often involve a mixture of public and private funding, whereby public funds are used to lever or supplement private funds. In general, this involves using public funds or interventions to improve the risk/return ratio of an investment, thereby potentially achieving greater impact than through solely publicly funded mechanisms. The policy interventions discussed below range from actions in financial markets (eg using public financing to reduce risks in investment funds) to supporting micro-enterprises who manage ecosystems (eg pro-biodiversity business models supporting family farms) with the aim of encouraging developments in biodiversity and ecosystem services markets. Environmental markets can also be driven by policy requirements; for example, compliance with regulations can involve market mechanisms that trade units of compliance such as carbon and biodiversity offsets markets.

There are potentially many finance mechanisms through which the conservation of biodiversity can be funded from private sources. Table 9-1 provides an overview of these instruments and brief description, developed from Dickie et al (2012). These have been grouped according to the most likely reason for private sector involvement in restoration, as identified above, ie philanthropic, profit-driven, or regulatory requirements. It should be noted however that there will be overlaps between the groups and some instruments could be placed in more than one. Private finance of Target 2 actions is likely to involve a combination of these mechanisms, working alongside public funding.

[149] Philanthropic intentions are taken in this study to encompass both altruistic and charitable intentions.

Table 9-1 Overview of the private sector financing instruments

Financing instrument	**Description**
Philanthropic donations	
• Not-for profit sources	Funds raised by organisations registered as non-profit for ecosystem management (ie that would otherwise have been spent on other issues), not including funds already earmarked for ecosystem management (eg by governments, companies or foundations) that are channelled via NGOs.
• Philanthropic donations by companies	Donations made by private sources that are not predicated on achieving a positive financial (or other private) return.
Profit driven investments	
• Payments for ecosystem services	A voluntary market mechanism in which suppliers are paid by beneficiaries to manage the ecosystems in such a way so as to enhance or continue the ecosystem service.
• Bonds for green infrastructure	A tradable financial security that promises to payback the holder at pre-defined interest rate used to fund projects with positive ecological impact. It can be delivered through a public private partnership in which government guarantees a certain level of payback to the private investor.
• Insurance sector mitigating of environmental risk	Using funds generated by insurance premiums to manage sources of environment risk such as natural disasters.
• Bio-carbon markets	The sale of carbon credits created through ecosystem maintenance or restoration that sequesters and/or reduces emissions of carbon.
• Pro-biodiversity business models	Investment or support structures for businesses that have a positive impact on biodiversity via a funding platform (eg NGO with restoration expertise).
• Product labelling and certification	Identifying a product to consumers whose production process supports a certain type of ecosystem maintenance and/or restoration.
• Tax relief on capital assets	Adjustment of tax rates to favour certain types of ecosystem management and maintenance of assets in good environmental management.
• Risk-sharing investment structures	The use of public sector guarantees to encourage private investment in ecosystem restoration activities that lead to business opportunities (eg loans on favourable terms, first loss on public shares or guaranteed minimum return on investment).
Regulatory measures	
• Hypothecated tax funds	Allocating tax revenues to specific spending objectives (eg through an environmental tax), potentially combined with requirements for private matched funding.
• Biodiversity offsets and habitat banking	A scheme whereby the losses of biodiversity as a consequence of a development or land use change are compensated by the investment in the restoration of another site of similar habitat, resulting in no net loss of biodiversity. Where offset outcomes exceed the damage done (net gain) they can contribute to ecosystem restoration.

9.3 Applicability of instruments to financing of Target 2

9.3.1 Suitability of the financing instruments

The different mechanisms are described in more detail and their suitability for financing Target 2 actions are assessed in Annex 3 against the following criteria:

- Suitability to attract private funding for activities under target 2.
- Private sector acceptability (both rate and timing of return).
- Financial scale compared to resources required to deliver Target 2 objectives.
- Spatial scale over which they are likely to function.
- Social equity and implications.
- Transaction costs to the public sector.
- Added value beyond good current management practice or offsetting of losses.

The main points from Annex 3 are summarised in the SWOT analysis below (Tables 9-2 to 9-4). Table 9-6 shows the possible opportunities for private funding for the maintenance and restoration of the habitats described in Section 6 of this report.

Table 9-2 SWOT analysis of philanthropic instruments

Mechanism	Strengths	Weaknesses	Opportunities	Threats	Conclusion
Not-for profit sources	Generates funds from the public, which can fund ecosystem maintenance and restoration. Knowledge and position in society allow NGOs to act as a trusted intermediary.	Different organisations (eg environmental NGOs) can duplicate functions. Limited to the resources they receive from donations and only sustainable as long as support continues.	Non-profit organisations can play a key role as intermediary in private finance of target 2, eg through pro-biodiversity business models and mediating the establishment of PES.	Economic downturn could result in less donations from donors. May have limited legitimacy with some sectors.	Non-profit sector generates some funding for ecosystem maintenance and restoration, but may make larger contribution as an intermediary in facilitating the other mechanisms to happen.
Philanthropic donations	A simple and direct mechanism and established as a means of funding ecosystem maintenance and restoration.	It is unlikely that funding from this source will increase significantly to make a contribution to achieving Target 2.	Donations can be supported through tax incentives. Environmental awareness could increase spotlight on corporate giving and CSR, encouraging donations.	Current weak state of the global economy likely to reduce donations, with an increased focus on the maintenance of profit margins.	Will support ecosystem restoration, but the contribution unlikely to increase significantly.

Table 9-3 SWOT analysis of profit driven investments

Mechanism	Strengths	Weaknesses	Opportunities	Threats	Conclusion
Payments for ecosystem services (PES)	Potential perpetual source of private finance with minimal government involvement. An approach that has been demonstrated throughout Europe.	Private use currently limited mainly to water services. There is a need to establish the benefits provided to a wider range of sectors in order to expand.	Emergence of stronger evidence on value of ecosystem services. Potential for expansion in water sector and/or schemes with multiple purchasers.	Public opposition to commodification of environment. Lack of knowledge of ecosystem services benefits from ecosystem restoration.	Well demonstrated for services tied to water, and could be expanded. Potentially limited by uncertainties over benefits of ecosystem restoration.
Bonds for green infrastructure	Can raise upfront finance needed to undertake ecosystem maintenance and restoration actions.	Needs to be attached to large scale activities with sufficient financial returns to justify formulating 'bonds'.	Long term returns on ecosystem restoration and stability of environmental assets (eg forests) suitable to bonds. Public sector could share risk.	Unclear if markets have appetite for bonds. Limited by uncertainty of payback of ecosystem restoration.	Potential mechanism to raise finance for target 2 actions organized on a sufficient scale and offering commercial returns if benefits are clear or risk shared.
Insurance sector mitigating environmental risk	A clear link exists between natural hazard risks and ecosystems, recognised already by some insurance companies.	May not fit within the institutional structure of the insurance sector. Cost-effectiveness of investments in ecosystems to mitigate natural risks uncertain.	An increase in natural risks (eg due to climate change) may increase the attractiveness of such mitigating measures.	If governments underwrite natural hazard risks, there is little incentive for the insurance sector to finance restoration.	Theoretical at present, further research and development needed before it is likely to be applied.
Bio-carbon markets	Existing carbon markets could be adapted to trade bio-carbon. Small part of carbon market could fund substantial ecosystem restoration and, to an extent, maintenance (see below).	Technical and institutional barriers to adapting carbon markets to include bio-carbon. Need to verify ecosystem restoration credits. Not all restoration generates carbon credits.	Defining clear standards for measuring bio-carbon credits from ecosystem maintenance and restoration would support market confidence and reduce transaction costs.	Weaknesses in global climate commitments undermining confidence in carbon markets.	Could be useful source of finance for Target 2 actions. Could mainly work in conjunction with other mechanisms (eg PES, labelling).
Pro-biodiversity business (PBB) models - funding	Intermediary between funding sources (public and private) and businesses (especially SMEs) that can deliver	Needs to be organized on sufficient scale to justify transaction costs. Opportunities where viable business models	Establishing PBBs where significant numbers of SMEs control an area suitable for commercial ecosystem maintenance	Uncertainties and conflicts in policies influencing ecosystem management (eg CAP).	Useful model to expand where significant numbers of SMEs can deliver commercial ecosystem maintenance

Mechanism	Strengths	Weaknesses	Opportunities	Threats	Conclusion
platforms	ecosystem maintenance and restoration actions.	also support biodiversity will be limited and site specific.	and restoration.		and restoration.
Product labelling and certification	Potentially huge market supporting maintenance (and incentivising restoration) of those habitats tied to provisioning services. Certification processes are well understood, reducing time needed for establishment.	Scope limited to ecosystems with commercial products. Price premium may not cover restoration costs. Challenge to communicate ecosystem restoration benefits through a label.	Defining new labels, or adjusting existing labels, that include ecosystem restoration in their labelling criteria.	Tough economic conditions could see consumers turn away from labelled products due to the cost. Proliferation of labels may confuse consumer.	A potentially large market to support Target 2. Requires consumer appetite for such information and products.
Tax relief on capital assets	Works through existing tax system and will encourage ecosystem restoration by those willing to invest in it for other reasons. Pay-off expected for large scale landowners (eg forestry, mining).	Difficulties persuading finance ministries to give tax breaks for the natural environment. Applies to restoration likely to have happened anyway. Targeting to restoration increases transaction costs.	Inheritance tax and taxation of land assets could be adjusted to support ecosystem maintenance and restoration actions.	Likely to come under pressure in time of fiscal constraints. Could be inequitable to those who already are maintaining or have restored ecosystems.	Potential to operate at a large scale eg forestry or mining sector, especially if already amenable to restoration. Signals strong policy commitment to Target 2.
Risk-sharing investment structures	Cost-effective use of public funds prompting private finance. Successfully used for energy efficiency.	Needs sufficient scale of potentially commercially viable ecosystem maintenance and restoration opportunities to invest in.	Risk-sharing structures could be used to support private funding through other mechanisms with best potential to support Target 2.	Uncertainty of extent and diversity of commercial activities that can operate alongside large-scale ecosystem maintenance and restoration projects.	Could play a useful role in facilitating investments for large-scale activities that otherwise would not happen.

Table 9-4 SWOT analysis of regulatory measures

Mechanism	Strengths	Weaknesses	Opportunities	Threats	Conclusion
Hypothecated tax funds	Works through existing tax system and levers private funds through matched funding. Gives fiscal incentive for ecosystem restoration actions.	Difficulties in persuading finance ministries to give tax breaks for the natural environment. Would apply to restoration that would have happened anyway. Targeting to restoration increases transaction costs.	Could direct private funds to support Target 2 through eligibility criteria and matched funding requirements.	Difficulties relinquishing tax revenues in time of fiscal constraints. Better if tax forgone has connection to ecosystem restoration.	Could lever new private funds to ecosystem restoration and would signal strong policy commitment to Target 2.
Biodiversity offsets and habitat banking	A strong no net loss policy would help maintain ecosystems by preventing loss and damage, and could help restore ecosystem if significant net gain occurs, especially if offsets are strategically located, eg to contribute to GI or ecological networks.	Offsets need to be strategically located and provide a net gain to go beyond maintenance and contribute to Target 2 restoration, but magnitude of net gains uncertain	Mandatory approach to NNL and offsets, strategic use of offsets (eg in support of green infrastructure priorities) and adoption of best practice would support Target 2.	Regulatory and philosophical barriers could restrict the development of this approach. Could result in net loss if inadequately regulated	Could support maintenance of ecosystems and some restoration if adequately regulated and if offsets deliver significant strategically located net gains, but this is uncertain

Distinction between the financing of maintenance, restoration and re-creation
The SWOT analysis highlights a subtle difference between the opportunities presented by the instruments for the funding of maintenance or restoration of ecosystems. In addition, the re-creation of ecosystems, ie restoration where the ecosystem has been destroyed in the restoration site, presents different challenges for financing instruments.

The distinction between re-creation and restoration is not always clear, and in general, the analysis of funding opportunities for both will be very similar. The main differences relate to higher up-front costs for re-creation in any given ecosystem, and possible longer times before financial returns are realised (ie because it would be expected to take longer to re-establish ecosystem services). Both of these reduce the prospects of making a financial return and therefore lessen the attractiveness of the investments to private funding sources.

An important factor to consider when distinguishing between these three types of measures is the degree of additional ecosystem service delivery provided by each. For certain instruments, such as PES and insurance sector funded mitigation of risk, the additional benefits provided by the measures are important; ie while re-creation may cost more, it might also bring higher ecosystem service gains and therefore higher financial return. Thus, re-creation could be important where the financing/returns reflect the scale of the ecosystem services restoration (ie PES, bio-carbon markets and donations by companies with the intention of burnishing CSR reputations). It will be less important for instruments that are solely dependent on achieving a certain level of ecosystem service (ie labelling, tax relief on maintaining assets in good condition) where re-creation will be less favoured compared to cheaper restoration and maintenance.

Maintenance measures are likely to attract less financing from those measures relying upon a high degree of visibility of the impacts of the investment (ie donations from companies, and hypothecated tax funds). Nevertheless, it is likely to work well for labelling, which relies upon the maintenance of a particular standard, such as quality of soils, forests or fisheries. It also holds an attraction for funding instruments reaping benefits of ecosystem service delivery (such as PES, bio-carbon markets or insurance sector funding) where the baseline scenario shows a deteriorating quality and an increase in future costs. In this case, maintenance may offer a cost-effective means of achieving a return on investment.

This raises the question of the relationship between ecosystem restoration/re-creation (assumed to be a continuum) on one axis and ecosystem service levels on the other. It is not possible to generalise except to assume that there can be high returns at the bottom of the restoration/recreation axis, and diminishing/low returns at the very top, but generalisations are unwise and different relationships can be observed.

Grouping of the funding instruments
The SWOT analysis and the in-depth analysis of each instrument (Annex 3) suggest that the mechanisms can be thought of in different groups with respect to how they operate. We identify ‘direct instruments’ which directly fund an activity on the ground and in each case tends to be specific to particular habitat type(s) which produces a desired service. The

application of some other instruments is less habitat-specific and these are described as 'enabling actions'. These are instruments that could be applied to any individual or groups of habitat types and can be used to improve the incentives to leverage private sector financing (see Table 9-5).

The role for enacting enabling instruments usually lies with a public body or a non-profit organization. Risk-sharing, public-private partnerships in delivering environmental bonds and tax incentives on capital assets are enabling actions which help finance viable commercial habitat restoration activities and are thus potentially suitable to support all types of private sector contributions to ecosystem restoration activities. Where the rate and/or of timing of return on investments in ecosystem restoration unattractive to private sector in a pure commercial sense, these instruments can be used to improve profitability. It differs to conventional uses of public spending, such as direct grants, in that funds are used to broker a deal to promote a commercial activity. The returns can be compared via ecosystem gain per public spend in return for a direct grant versus ecosystem gain from the overall commercial activity per public spend.

Table 9-5 Types of innovative financing instruments

Instrument type	Description	Examples
Direct instruments	Result in a direct change on the ground; are specific to particular ecosystem types.	• Payments for ecosystem services • Product labelling and certification • Bio-carbon markets • Biodiversity offsets and habitat banking • Insurance sector mitigating of environmental risk • Bonds for green infrastructure • Pro-biodiversity business models
Enabling instruments	Support actions on the ground but do not directly guarantee them; are applicable to any group of habitat types. Usually used to improve the incentives for ecosystem restoration.	• Philanthropic donations by companies • Tax relief on capital assets • Hypothecated tax funds • Risk-sharing investment structures • Not-for profit organisations contributions (can also act as an enabling instrument)

In addition, a number of key themes emerged from the analysis. Firstly, there is a clear advantage for those instruments which are already close to commercial viability, ie 'pay their own way' in terms of the returns they are able to accrue for their investors, for example, through PES or reduced risk. This reduces susceptibility to changes in government policy.

Secondly, mechanisms that are already well understood and/or already operational have a much greater chance of rapid implementation. A considerable barrier to a number of the instruments is the lack of understanding or experience of investors with these models. This applies both to the uncertainty of returns and lack of understanding of how they will function in practice. Uncertainty increases risks for private sector investors. Thirdly, there is a need for clear markets to exist to provide a reliable trading platform and drive demand.

A number of mechanisms are identified as the best options for several habitats in Table 9-6, suggesting that they offer more promising options for private finance of Target 2 actions.

Table 9-6 Examples of the use of innovative funding of measures under Target 2 of the Biodiversity Strategy

Target 2 habitat (habitat-specific actions)	**Innovative direct instrument**	**Examples of key socio-economic benefits/services provided**	**Supporting public funds, policies or enabling instruments**	**Example / Case studies**
Farmland: arable, permanent crops, and temporary & improved grasslands.	Payments for ecosystem services	• Water purification and regulation	• Agri-environment	Payments by water companies to farmers for improved farm management and restoration practices resulting in lower water treatment costs (eg Vittel and SCAMP case studies – see Annex 4).
	Product labelling and certification	• High nature value farming • Cultural & landscape values	• Agri-environment • Tax relief on capital assets • Risk sharing investment structures	
Natural and semi-natural grasslands	Payments for ecosystem services	• Water purification and regulation	• Agri-environment	Payments by water companies to farmers for improved farm management.
	Product labelling and certification	• Provision of food from HNV farming • Cultural and landscape values	• Agri-environment • Tax relief on capital assets • Risk sharing investment structures	Maintenance of species rich grasslands allows the production of certain products (eg Ennerdale beef using extensive grazing by hardy native breeds).
	Bio-carbon markets	• Carbon sequestration	• Agri-environment • Bonds for green infrastructure	Wet grasslands managed to maintain hydrological conditions eg via maintenance of dams/sluices.
	Pro-biodiversity business models	• Provision of materials (eg straw) • Provision of food	• Agri-environment • Establishment of financing platforms.	Use of products of mowing / cutting / hay making.
Forests	Product labelling and certification	• Timber • Cultural and landscape values	• CAP Pillar 2 payments for forestry.	Payment of premium to ensure the sustainable harvest and management of forests (eg FSC).
	Insurance sector mitigation of risk	• Erosion regulation • Flood protection	• CAP Pillar 2 payments for forestry. • Bonds for green infrastructure	
	Bio-carbon markets	• Carbon sequestration	• Bonds for green infrastructure	
	Pro-biodiversity business models	• Timber (and other wood products)	• CAP Pillar 2 payments for forestry.	
	Payments for ecosystem services	• Erosion regulation; • Flood protection	• CAP Pillar 2 payments for forestry.	

Target 2 habitat (habitat-specific actions)	Innovative direct instrument	Examples of key socio-economic benefits/services provided	Supporting public funds, policies or enabling instruments	Example / Case studies
Heathland and tundra	Product labelling and certification	• Livestock and related products • Game/hunting	• Agri-environment • Tax relief of capital assets	Low intensity grazing could result in biodiversity friendly animal products. Eg Wildlife Estates certification (see Annex 4).
Sclerophyllous vegetation	Insurance sector mitigation of risk	• Fire prevention	• Agri-environment (in certain Member States). • National forestry payments	Livestock grazing maintains the habitat from succession while also reducing fire risk. Very little funding afforded to this habitat as it does not normally fall under the definition of grasslands.
Mires (bogs and fens)	Bio-carbon markets	• Carbon sequestration	• Agri-environment	Re-wetting of bogs results in both sequestration and reduced future carbon emissions.
	Payments for ecosystem services	• Water purification and regulation • Erosion regulation	• Agri-environment • Reduction of regulatory barriers	Removal of colour and pollutants, as well as holding water during drier months. Payments by water companies for improved farm management and restoration practices (eg Vittel and SCAMP).
Inland marshes	Payments for ecosystem services	• Water purification	• Agri-environment	
Freshwater ecosystems	Payments for ecosystem services	• Water purification and regulation	• Agri-environment	
Coastal ecosystems: beaches, dunes, saltmarshes, estuaries lagoons	Payments for ecosystem services	• Water purification	• Agri-environment	Dunes have been shown to improve water purification (eg €86m/yr compared to €3.2m/yr for management) (Meulen et al, 2008).
	Insurance sector mitigation of risk	• Flood protection	• Agri-environment	Salt marsh restoration can reduce costs of maintaining sea defences (Rupp and Nicholls, 2002)
	Bio-carbon markets	• Carbon sequestration	• Bonds for green infrastructure	Coastal wetlands and marshes
Marine ecosystems	Product labelling and certification	• Capture fisheries • Aquaculture products	• Common Fisheries Policy measures to support sustainable fisheries.	Catching of fish from a stock that is sustainably managed. Eg MSC certification has now close to 8% of the total of the total wild capture harvest.[150]

[150] http://www.msc.org/business-support/key-facts-about-msc

On the basis of these findings, we have selected three instruments that appear to have the greatest potential for delivering large scale restoration in the near future: payments for ecosystem services, product labelling and certification, and bio-carbon markets. These are explored in more detail below and the following section.

Payments for ecosystem services (PES)

Payments for ecosystem services are regarded as a good prospect as they are already an established mechanism. Public-sector PES through agri-environment schemes is relevant to the management of farmland, forests, grasslands, dunes and mires. The existence of these PES arrangements provides a mechanism through which payments from private sources for ecosystem services can be made. This is already happening in the water sector, where private payments may be combined with public funds (see SCAMP and Vittel case studies).

These water sector PES mainly address diffuse pressures on water resources (eg water colouration or nutrient pollution), but usually do so through restoration of farmed habitats. They can therefore directly lead to restoration of several terrestrial habitats and to relieving pressures that are a barrier to restoration of water bodies, including coastal wetlands. The expansion of PES is expected to continue in the water sector, but has potential to be expanded to other industries with strong reliance on water supplies. This is more likely in PES arrangements where multiple purchasers can be found.

Coastal wetlands provide other ecosystem services that could be traded through PES, such as carbon storage (see bio-carbon below) and supporting fisheries productivity. Its recognition in public funding for fisheries management could lead to private support for coastal ecosystem restoration. This is more likely for less mobile commercial species, such as shellfish.

Product labelling and certification

Labelling products as environmentally-friendly in order to attract a price-premium from customers is already a well-established approach for channelling private funding into ecosystem restoration and management. Organic and high-nature value labels already exist in a variety of agricultural systems used to manage farmland, grasslands, moorland and heath. Markets for timber and fish that are sustainably produced are also well-established (eg Forest Stewardship Council (FSC) or the Marine Stewardship Council (MSC)).

However, there are limits to the financial scale of these approaches, both in terms of the size of the price premium that customers are prepared to pay and the extent of the market (number of customers). Therefore, they often require some public funding (eg as in the additional agri-environment payments for organic farmers).

Environmentally-labelled production processes may not always deliver Target 2 objectives for the ecosystems involved. Nonetheless, they can be expected to make a significant contribution where they are used. For example, Marine Stewardship Council fisheries certification relates to the sustainable exploitation of particular commercial fish species, but also has some requirements to control fisheries pressures in marine areas, contributing to ecosystem restoration.

Bio-carbon markets
Private purchase of carbon credits, although much smaller than regulated emissions trading arrangements, is already one of the largest private-sector driven environmental markets. Part of this market already involves purchases of carbon credits from actions that are also positive for biodiversity (bio-carbon). This market has potential to expand, with the most likely habitats to benefit being those whose management has the greatest impact on carbon emissions. Forests are the most widespread relevant habitat, but mires, peat soils and coastal wetlands have also been identified as significant stores of carbon.

Most promising of enabling instruments
Some other options in Table 9-6 are regarded as of cross-cutting relevance to Target 2 actions. The most promising of these are use of the tax system and encouragement of philanthropy.

Taxation is one of the most influential ways that governments intervene in markets and therefore influence use of private sector resources. Two mechanisms considered in this section (tax breaks and match-funded hypothecated taxes) provide options for using the tax system to encourage private funding of Target 2 actions.

Their appropriateness for delivering Target 2 is dependent on existing Member States' national tax structures, land ownership structures and the main ecosystem restoration and maintenance priorities. For example, in a Member State with mainly private land ownership where maintaining ecosystems is the dominant challenge, inheritance tax breaks may provide the best incentives. In a Member State with existing resource-use taxes (eg on non-renewable resource extraction) and more extensive actions needed to restore ecosystems, allowing hypothecation of taxes with private matched funding into ecosystem restoration may be a more suitable instrument.

Encouraging philanthropic funding for Target 2 actions is also highly dependent on Member State circumstances, as most will already have existing systems for encouraging charitable giving. The best actions may therefore be to adapt these to give greater encouragement to ecosystem restoration, or to simply highlight the benefits of ecosystem restoration. Greater awareness of the benefits of ecosystem restoration could motivate more philanthropic support, in particular, from the CSR budgets of a wider range of large private companies with links to ecosystem management.

While tax-system and philanthropic measures can be encouraged at a European level, their implementation is most likely through Member State actions.

Combining public and private
The case studies (see annex 4) mention several examples of combining public and private resources to support Target 2 actions. In these cases the way that public sector resources are used can be adapted to encourage greater private sector funding. In financial markets, this can involve interventions that reduce investment risks, through public sector taking on a share of risk, an approach that is already widely used to encourage private sector activities (eg research) in other policy areas (eg energy efficiency).

In order to channel private resources into Target 2 actions, new funding platforms or processes may be required, in line with approaches recently explored through pro-biodiversity business models. A funding platform is an organizational arrangement through which expenditure (possible from multiple sources) is channelled to particular objectives. An example would be a research fund for a particular sector or area of science.

Box 9-1 Use of agri-environment scheme with PES

In the SCAMP PES scheme in the UK, the commitment of 10 years of public funding of agri-environment schemes (AES) was made alongside water company capital investments. The return to the water company was too uncertain to have stimulated this capital payment without the guarantee of AES funds, and similarly the AES funds alone where not high enough to incentivise farmers to restore and maintain the catchment. However, in combination the public and private funds were sufficient to make the PES scheme financially viable. Close co-operation and a good working relationship between the actors was seen as an important factor in developing the initiative.

Source: P. Austin, SCAMP manager, United Utilities, pers. comm.

9.3.2 Potential scale of financing instruments

A key factor in deciding which measure from those listed above should be adopted to encourage private finance of Target 2 actions is the scale at which they could operate. A summary of the amounts of funds raised in the case examples reveals that significant amounts of private sector financing has been found in individual cases, including through payments for ecosystem services. Those with the greatest amounts of funds raised are those in which measures are introduced at the national level to either leverage funds from the private sector (eg Dutch Green Funds) or act as an environmental tax (eg UK Landfill Levy). However, both of these require the relinquishing of tax revenue for restoration purposes or the imposing of an environmental tax, which are likely to find little political support in times of austerity.

Table 9-7 Amounts of funds raised in the identified case studies

Instrument type	Case example	Funds raised (€)	Description
Philanthropic donations by companies	Moorland Protection Fund, Germany. Funded by Volkswagen.	€1.6 million (Ongoing)	This sum refers to those raised by Nov. 2011.[151] Donation of fixed amount for each vehicle leased.
	LaFarge (mining company) floodplain rehabilitation.	€80,000 (2009 – 2011)	100ha restored, supported by WWF.
Payments for ecosystem services	Vittel water funding of Water Catchment Management, France.	€24 million (1993-2000)	Includes land acquisition (€9m), compensation (€11m) and farm equipment (€4m).
	‘Drinking Water Forest’ funded by Bionade to offset water use, Germany.	€1 million (2008-2011)	Over 60ha of deciduous forest planted plus upkeep. 100,000 m^3/yr of water regeneration.
	Sustainable Catchment Management Plan (SCAMP), UK. United Utilities	€16 million	Restoration of 55 km^2 of blanket bog; 4.5 km^2 of upland

[151] http://www.volkswagenag.com/content/vwcorp/info_center/en/news/2011/12/fonds.html

Instrument type	Case example	Funds raised (€)	Description
	funding of upland bog management.		oak woodland.
Product labelling & certification	Wildlife Estates label, industry led certification for hunting estates managed in biodiversity-friendly manner.	Undisclosed	Estates to halt and reverse the loss of biodiversity.
Risk-sharing investment structures	Verde Ventures debt financing scheme for small and medium enterprises (SMEs).	€15 million	Currently only outside the EU. Protection or restoration of 4,640 km^2. €102 million sales of eco-friendly goods/services.
Tax relief on capital assets	Dutch Green Funds: tax compensation provided by the government for individuals who invest in green institutions below market returns.	€12 billion (1995-2011) (€750 million per year)	Biodiversity measures only a small proportion. Two thirds directed to energy efficiency measures include organic farming and 1,250 km^2 of restoration. Government invested approx. €150m.
Hypothecated tax funds	Landfill Communities Fund, UK. Restoration actions funded by a part of the Landfill Tax.	€59 million (1993-2012) (€5.9 million per year)	Ongoing. Total fund has raised €1.5 billion over this period.
	Aggregates Levy Sustainability Fund, UK.	€61 million (2002-2011) (€6.1 million per year)	Constitutes around 7% of the total sum paid by aggregates companies (ca €370 million).
Biodiversity offsetting/ habitat banking	CDC Biodiversité pilot offsets scheme, France.	€13 million	Approx. €3,800/ha.

Source: See case studies in Annex 4

Scalability is considered in annex 3, but is further analysed here with reference to the three instruments considered of particular interest. In this analysis, two aspects of scalability are considered for the measures discussed below:

i) The area of ecosystems that could potentially be covered, and
ii) The proportion of the costs of Target 2 actions that could potentially be financed by private sources.

Payments for Ecosystem Services

<u>Area of ecosystems they are suitable for</u>
PES schemes broadly occur where one party constrains the provision of an ecosystem service that is valued by a second party. This limitation will occur through an intervention on the ecosystem. Those industries that physically manipulate ecosystems and have the potential to change are theoretically suitable for PES, for example agriculture, agro-forestry, resource extraction, fisheries, construction, tourism or manufacturing. But the application of PES has mainly been demonstrated in agriculture and agro-forestry. For this reason PES is only likely to operate at present for ecosystems that have some provisioning ecosystem services (eg producing food or fibre).

What portion of Target 2 costs can they cover per ha?

Agri-environment schemes have been classified as the largest PES scheme in operation. Agri-environment payments provide an indication of the scale of payments required to implement a functioning PES scheme. As an example, payment rates from the UK's agri-environment scheme are shown below:

Table 9-8 Payment rates from the UK's agri-environment scheme

Scheme	Standard Payment (2013)	Management options payment
Entry Level Scheme[152] (ELS)	37 €/ha/yr	Management actions come with points. Points target much be reached for ELS. Some supplementary action available at €6 per ha.
Uplands ELS	76 €/ha/yr for severely disadvantaged areas	Points target much be reached
Organic Entry Level Scheme (OELS)	74 €/ha/yr	Points target much be reached
Uplands OELS	113 €/ha/yr	Points target much be reached
Higher Level Stewardship (HLS)[153]	Dependent on management options	Ranging from 6 to 862 €/ ha.

It is likely that widespread PES schemes (like ELS) will not provide high enough payments to cover many of the actions that are required for restoration targets. For example, they fall well short of the estimated cost of restoring and maintaining degraded grasslands (of 1,750 €/ha/yr).

Under the Vittel scheme (see case studies, annex 4), farmers were paid 200 €/ha/yr to join the scheme. This annual payment was supported through capital investment on each farm, of approximately €150,000 per farm. The total cost for the scheme operating between 1993 and 2000 was approximately 980 €/ha/yr. This level of payment could be sufficient to fund restoration actions, as once an allowance for inflation between 2000 and 2012 is made, it covers a large portion of the figure provided above required for grasslands.

Table 9-9 Estimated cost according of maintaining and restoring 15% of degraded grassland ecosystem according to overall degradation levels and the costs of combined measures in 2020 (see chapter 6.4)

9.4 **Cost type**	**€/ha/yr**
Maintenance cost	250
Restoration Cost	1500 €/ha/yr

How suitable they are to being used at a large scale

PES is a description of a transaction between two bodies (often facilitated by an intermediary such as an NGO). It is not prescriptive in the scale over which it can take place.

[152] NE349: Entry Level Stewardship: Environmental Stewardship Handbook, Fourth Edition - January 2013
[153] NE350: Higher Level Stewardship: Environmental Stewardship Handbook, Fourth Edition January 2013

PES exist at the international scale, such as in the Clean Development Mechanism. But they also can take on a local basis such as the Vittel example in North East France or SCaMP in the North West of England. At present the majority of companies that are willing to pay for PES in Europe are related to water. The impacts of land use upon water quality is well understood and there are large companies with a stake in the attributes of water who are able to act strategically to influence the way that land is managed in their catchment, resulting in a potentially large areas of land being suitable for PES schemes.

Product labelling and certification

Area of ecosystems they are suitable for

Certification schemes and their labels benefit landowners/manufacturers by attracting consumers to their products. These products tend to come from managed ecosystems where overexploitation or poor management can cause damage. Product labels are a sign that the actions a business is taking to minimise their negative impact or maximise their positive impact on the environment in the course of their business activity, and that this has been approved by an independent body.

Certification can be applied to any ecosystem where a business relies on an ecosystem service to produce a market product. This can be where the reliance is immediate, for example farms, tourism or recreation, or further down the supply chain in processed foodstuffs or clothing where the sourcing of certified raw materials is important.

The most obvious application of certification opportunities lie in productive ecosystems, where a product is to be sold directly to a consumer or business. Examples of certification schemes and the ecosystems they protect are given below:

- Organic certification[154] covering a range of agricultural ecosystems.
- The Marine Stewardship Council (MSC) certification[155] will ensure certain fisheries are managed in a certain way which will have helped protect marine ecosystems.
- The Wildlife Estates Label[156] protects a wide variety of ecosystems that support recreational hunting and fishing.

What portion of Target 2 costs they can cover per ha

Certification schemes require certain actions to promote biodiversity. On farmland such actions may make a farm eligible for Environmental Stewardship Scheme payments. If a farm was to turn to organic production it is likely to qualify for the Organic Entry Level Stewardship Scheme (OELS). The presence of public subsidy payment suggests that price premiums for organic production are at present insufficient to cover the costs of the ecosystem restoration and maintenance actions involved in establishing organic production.

Details of these payments are provided in the above section. Additional financing comes through the price premium that certified products can attract. These price premiums vary according to the product and the country in which it is sold. A price premium doesn't necessarily increase the profitability of a business as less intensive management may reduce

[154] http://www.soilassociation.org/certification
[155] http://www.msc.org/
[156] http://www.wildlife-estates.eu/

yields, but may help meet the costs of ecosystem restoration and/or maintenance. In simple terms the additional costs of Target 2 must at least be equal to a combination of any subsidy and the revenues generated from sales of certified produce. The returns generated in this way are in competition with those achieved by alternative uses for the land.

Table 9-10 The price premiums attracted for various conservations schemes

Certification Scheme	Price Premium
FSC	12%-20% (NB. for tropical sawn hard woods)[157]
MSC	14.2%[158]
Conservation Grade [159]	9 €/t for milling wheat[160]. Assuming organic yields of 7 t / ha[161]. Therefore €63 per harvest.

It is assumed that these price premiums cover the additional costs of producing and supplying a certified good. If this were not the case, certified goods would not exist. The key question for Target 2 is whether the actions as required under a certification scheme can contribute towards it. This can happen if a certification scheme is already supporting ecosystem restoration and maintenance, or if additional requirements could be added to certification criteria to contribute to Target 2. In the latter case, the additional actions would need to be covered by the price premium achieved for the product. A final option is that new certification schemes are set up that support produce from systems that manage ecosystems in a way that contributes to Target 2.

To estimate this in detail would require extensive analysis. However, the indicative price premiums presented above suggest that this may be possible from current levels of production.

<u>How suitable they are to being used at a large scale</u>
It must be recognised that if this mechanisms was to contribute to additional ecosystem restoration and/or maintenance, this would mean the area of certified production would increase. This would be expected to decrease prices unless there was an equivalent increase in demand. The price premium required to support many certified products may limit this demand. Many certification labels are currently in existence and therefore certification schemes have potential to operate at a large scale. For example, at the end of 2010, 10 million hectares were under organic production in Europe.[162] In addition, the UK saw expenditure on sustainable fish increase from £70m to £178 million between 2007 and 2009 (154% growth) despite growth in overall household expenditure of just 1%[163].

[157] https://www.pwc.com/en_GX/gx/sustainability/executive_summary.pdf
[158] http://www.msc.org/newsroom/news/new-study-shows-uk-supermarkets-secure-price-premium-for-msc-labelled-seafood-products
[159] http://www.fwi.co.uk/Articles/20/06/2012/133493/Cereals-2012-Demand-fuelling-conservation-grade-contracts.htm
[160] http://www.fwi.co.uk/Articles/20/06/2012/133493/Cereals-2012-Demand-fuelling-conservation-grade-contracts.htm
[161] http://www.conservationgrade.org/wp-content/uploads/2010/09/environmentalscientistapr10.pdf
[162] http://www.fibl.org/en/themen/themen-statistiken.html
[163] http://www.goodwithmoney.co.uk/ethical-consumerism-report-2010

Bio-carbon markets

Area of ecosystems they are suitable for

The ecosystems that bio-carbon markets will operate within must protect or enhance biodiversity and sequester carbon. The schemes must operate on ecosystems that are not likely to be disturbed, to ensure lasting sequestration and protection. For these reasons the ecosystems that are most suited to bio-carbon markets are the restoration of forests, peat bogs and coastal wetlands.

What portion of Target 2 costs they can cover per ha

Voluntary bio-carbon offsets or credits require payment to an organization in return for a volume of carbon emissions reduction. It is unclear as to the rate at which this payment is converted into areas of land. This conversion rate will depend on two key factors:

- Sequestration rate on the restored land – if the land sequesters a lot of carbon per ha, it can offset greater emissions and therefore attract a higher price.
- The opportunity cost of the land – in purchasing land or requiring a land owner to undertake a certain set of actions, the land owner's opportunity cost must be covered. A bio-carbon offset is more likely to operate from an ecosystem where these opportunity costs are low, and therefore providing it with a competitive offset price.

The World Land Trust offers carbon offsets, charging 18.50 €/CO2 ton (as of December 2012). An example of the amount of carbon being offset from a German peatland restoration project is that it saves 10 tonnes of avoided carbon emissions per ha per year (based on 29,764 ha being restored in Germany saving approximately 300,000 t CO2 equivalent per year[164]). Offset payments could therefore fund restoration of this peatland worth 185 €/yr. By comparison the opportunity cost of restoration of a peatland can be 585 €/ha/yr, plus foregone subsidies which can exceed 300 €/ha/yr [164]. Lower figures for restoration are provided in a study looking at the potential for Irish peatlands for carbon uptake and storage[165]. This study shows that restoring birch woodland on peatland costs 120 €/ha and restoration of alkaline wetland costs 250 €/ha. These figures are closer to those revenues achieved through the World Land Trust offsetting scheme.

How suitable they are to being used at a large scale

Bio-carbon markets already exist and have a proven application. The demand for this voluntary mechanism is driven by the desire of business to engage in CSR activity and individuals to offset the carbon impacts of their energy use. The voluntary Carbon Offsetting market is a large environmental market with US$576 million worth of transactions occurring in 2011[166]. This demand is supported in part by offsetting flights and CSR activity by corporations. In the UK, the Wood Carbon Code has protected 1 million tonnes of CO2 from the atmosphere since its launch, covering 2,733 ha[167].

---

[164] http://www.eea.europa.eu/atlas/teeb/peatland-restoration-for-carbon-sequestration-germany-1/view
[165] http://www.epa.ie/downloads/pubs/research/climate/CCRP_15_web.pdf
[166] http://www.clickgreen.org.uk/analysis/business-analysis/123577-carbon-offset-market-hits-three-year-high-following-strong-us-demand.html
[167] http://www.forestry.gov.uk/newsrele.nsf/AllByUNID/5761F6AF454A6940802579C10051ACF0

Although opportunity costs are potentially lower in the developing world, biocarbon markets could be encouraged to develop in Europe.

Scale of operation of enabling actions

Given their motivations, NGOs and foundations could make an important contribution to financing restoration projects where these appear to deliver high benefits to biodiversity but have more limited prospects as regards financial return on investment. The contribution can be substantial. For example, Morling (2008) estimated that UK NGOs spend £144 million domestically and £15 million overseas on biodiversity, constituting between 39% and 75% of the UK Government's respective domestic and overseas spend.

Risk-sharing actions need to work through financial market activities. This means they can only work at a relatively large scale, at which the application of financing instruments is worthwhile: for example, the EIB refers to investments of <€10 million as 'proof of concept' (ie full-scale pilots). That may be appropriate for some ecosystem restoration actions (eg large GI-type restoration investments) but is not the usual scale at which ecosystem restoration activities are organised. Ideally risk-sharing instruments will support ecosystem restoration investments as a package across geographical areas and ecosystem types, as this spreads risks (which it is in the public interest to do as a way of supporting ecosystem restoration investments). Their unit costs can be relatively high for the environment sector, but involve a large element of fixed costs, so the key for reducing them to a smaller percentage of overall spending is to make the investment arrangements of a sufficient size.

The instruments need to be considered to have different roles; and the challenge will be to find where to use different blends of the instruments. For instance, a European Commission project has explored the potential to combine commercial loan funding with public subsidies to establish 'Pro-biodiversity businesses' which maintain or enhance biodiversity (RSPB, 2010). It demonstrates a potential to establish profitable SMEs which create employment in rural areas, potentially gaining revenue from public payments in return for providing public ecosystem services and sale of biodiversity-friendly products and services into private markets.

Summary on scale of measures

The discussion of scale distinguishes two dimensions: the proportion of the unit costs of ecosystem restoration per hectare of habitat that the mechanisms could fund and the area of the ecosystem across which it could be applied. To be an instrument making a major contribution to Target 2, a high percentage is needed in both of these categories.

The three instruments assessed in detail can only be applied to certain habitats. Both labelling and PES can potentially be used at a large geographical scale (some examples of them already are). However, their potential to cover the unit costs are considered less strong, due to the uncertainty in the price premium consumers will pay for labelled goods, and the strength of demand for ecosystem services from actions to deliver Target 2.

Bio-carbon markets have potential to fund restoration actions and be applied to several important ecosystems (but not all ecosystems). However, carbon demand is still uncertain, no least due to regulatory risk.

9.5 Conditions that enable development of private sector financing

Private sector financing of biodiversity is already taking place, as demonstrated by the case studies. In some cases, the motivation is primarily the securing of profits in the long term (eg the Bionade payments for securing water) or the development of new markets (such as the FSC). In other cases, certain governments have introduced targeted environmental taxes, the proceeds of which they have invested in restoration initiatives or used to further leverage private sector funds (eg the Aggregates Levy Sustainability Fund in the UK). However, at a European scale, these measures have yet to make a significant contribution to funding restoration measures.

The reason for this may be that biodiversity suffers from a number of market failures:

- Information failures: the lack of understanding of the process by which it supports fundamental services.
- Many of the services provided by ecosystems are public goods, meaning that they cannot be solely owned or traded by a single party.

A number of the instruments rely upon the ecosystem services return provided and therefore are potentially limited by uncertainties over benefits of ecosystem restoration. These instruments include PES, environmental bonds, insurance sector mitigation of risk and bio-carbon markets. In many cases, it is not clear to what extent restoration in particular will in fact result in the return of desired services; this lack of clarity is limiting investment in these measures. A number of generic conditions are identified to facilitate the scaling up of private sector financing of biodiversity.

<u>Improve the levels of trust between the investors and key stakeholders</u>: a lack of understanding and trust between investors, policy makers and stakeholders on the ground has been cited commonly as a significant barrier to scaling up these schemes (Cranford et al, 2011). Such an approach has been taking place under the auspices of the Global Canopy Project to promote Forests Bonds in tropical rainforest. In many good-practice examples, trust has been built through relationships established via NGOs.

<u>Stability of policies:</u> investors must be confident that the regulations enacted by governments (such as placing a price on biodiversity protection) will be upheld by subsequent governments (Dickie et al, 2012). Cross-party consensus on how to engage the private sector is therefore required.

<u>Burden sharing between public and private sectors</u>*:* Financial viability is most likely to be achieved through a combination of private and public funds. Burden sharing between the public and private sector to cover these benefits might encourage private sector financing. A number of examples include:

- The use of flood defence budget in public sector environment agencies to co-fund some of the schemes that might appeal to insurers. This can also help tackle the investment uncertainty related to the uncertainty in environmental responses (as described above).
- Green infrastructure bonds should hold an investment grade credit rating. Public policy can create a price signal to stimulate early investment in ecosystem or green infrastructure

Start where goodwill already exists: for instance, Cranford et al (2011) suggest that with respect to forest bonds, the first bonds should target those with a socially responsible investment mandate already in place.

In addition to the above, there is the need to develop flexible regulatory frameworks: ie do not prevent businesses from financing ecosystem maintenance and restoration, and be able to adapt regulatory requirements. An interesting example of this was provided by the SCAMP example in northern England (see Box 9-2).

Box 9-2 The benefits of a flexible regulatory environment, SCAMP UK.

In the United Utilities payments for ecosystem scheme, known as SCAMP, the national regulator facilitated investments made by the private company by being flexible on their rules of financial reporting. Currently, all expenditures must be reported under either 'revenue', which means it is attributed to the annual accounts and affects the profit margin of that year, or as 'capital costs' for expenditures that represent a long-term investment, the costs of which are spread over longer time periods. As United Utilities owned the land on which the work was carried out, Ofwat facilitated the project by allowing Unitied Utilities to report expenditure under 'capital costs', with the understanding that they constitute a long-term investment, which allowed them to apportion the costs of the agreement over several decades. This was a factor in allowing the scheme to go ahead and is now being used by other water companies in the UK.

Conversely, any expenditure for works on land not owned by the company is currently reported as 'revenue', which affects the profit margin, making it a barrier to expansion. Nonetheless, Ofwat recently allowed a signed agreement between a water company and a neighbouring farmer to itself become an asset, which meant the costs could be reported under 'capital costs'. Allowing this practice to become more widespread practice in the future present the opportunity to expand this kind of program to land not owned by the water company. For example, the water company may decide to buy a new system for a neighbouring farmer to reduce pesticide application, which could save water treatment costs several times greater than the upfront costs. However, it may create incentives for bad agricultural practices in order to receive compensation to change.

Source: United Utilities SCAMP manager, pers. comm.

Establishment of a common framework for the measuring and reporting of the environmental and social impacts should be developed and tested (such as carbon or biodiversity) (Petley, 2012).

- There is considerable uncertainty as to the extent that ecosystem restoration may contribute to many ecosystem services (eg natural hazard regulation) thus deterring potential investment from the private sector such as insurance companies and, in all likelihood, public funding.
- A good practice standard could be developed by national government to guide and certify those schemes meeting specified standards.

Box 9-3 Policy review for ecosystem approach – the UK Natural Capital Committee

The UK Government has recently established a natural capital committee, whose remit includes identifying natural capital that is being used unsustainably. A policy structure for considering this question can support efforts to restore ecosystems. It brings together the need for analysis of current and future levels of services being supported by ecosystems and the consequences for human welfare of deteriorating and/or low levels of services. This in turn can provide economic information on the benefits of ecosystem restoration, allowing better judgement of where the costs of restoration are justified by the benefits. See http://www.defra.gov.uk/naturalcapitalcommittee/.

10 CONCLUSIONS

10.1 The costs of maintaining and restoring ecosystem to achieve Target 2 of the EU Biodiversity Strategy

This study has estimated the additional costs of achieving Target 2 (ie additional to the costs of existing measures and others that are expected to be taken up to 2020 under a reference scenario). Three main sources of cost were assessed, those relating to:

- supporting actions listed in the Biodiversity Strategy;
- measures that treat the source of generic wide-scale pressures (such as water and air pollution) ; and
- practical ecosystem management, restoration and re-creation measures (such as water management, grazing and removal of invasive species).

As these types of action are all differ considerably in many ways their costs were assessed separately, and are briefly summarised below.

10.1.1 The costs of supporting actions listed in the Biodiversity Strategy

As discussed in Chapter 0 the achievement of Target 2 will be dependent on a number of interacting targets and actions under the Biodiversity Strategy. However, the following three supporting actions (and associated sub-actions) listed under Target 2 are of particular relevance, and there cost were therefore assessed in this study:

- Action 5: Improving knowledge of ecosystems and their services.
- Action 6: Promoting a green infrastructure.
- Action 7: Ensuring no-net loss of biodiversity.

Given the nature of these actions and information available, it was not possible to carry out a quantitative or in-depth analysis of their associated costs. Furthermore, some of the actions are still being further defined, or were only recently completed before the completion of this study. As a result the cost assessments should be considered to be of a preliminary nature.

Despite these limitations on the analysis, it is evident that as most of the actions are primarily research, strategic planning or promotion measures then their costs (probably in the 10’s of millions of Euros in total) will be insignificant compared to other more practical land management and restoration measures that will need to be taken to achieve Target 2. In fact, given the very high costs of practical measures, such studies will have a major role to play in strategically planning and ensuring the best use of resources for practical measures, and may therefore be expected to be vital cost-effective components of the Strategy.

Although the implementation of the Commission’s recently published Green Infrastructure Strategy will undoubtedly require some land management measures with associated significant costs, these are considered in the assessment of practical ecosystem-specific costs. To avoid double counting they are not therefore assessed and included in Green Infrastructure costs.

10.1.2 Measures that treat the source of generic wide-scale pressures

It is evident from a number of monitoring programmes and research studies (as referred to in chapter 2) that some of the most widespread and significant impacts on ecosystems and their associated species result from wide-scale water and air pollution related pressures. These pressures will therefore need to be tackled if Target 2 is to be achieved. However, such pressures cannot normally be easily dealt with at their point of impact, but generally require measures that reduce pollution emissions at their source. This study therefore examined the implementation of existing measures that are being taken to reduce pollutant emissions and attempted to establish if such measures would be sufficient to achieve Target 2. However, as such measures are wide ranging and aim to tackle a range of environmental problems it was not possible to ascertain the proportion of costs that might be attributed to achieving Target 2. Furthermore, as sophisticated modeling would be required that is beyond the scope of this study, it was not possible to estimate additional costs that might be entailed to address any shortfalls in measures with respect to the achievement of Target 2.

In conclusion, it appears that existing policy and legislative measures, most notably the Urban Waste Water Treatment Directive, Nitrates Directive and more recently the WFD provide strong legal frameworks to achieve ecosystem restoration and maintenance for Europe’s waters. Moreover, requirements under the WFD to prevent the deterioration of waters and to achieve Good Ecological Status or Good Ecological Potential appear to be equivalent to Target 2 requirements for maintenance and restoration. Although there have been some implementation difficulties with the WFD, and Member State reporting indicates that progress is slower than expected, it seems likely that Target 2 restoration requirements related to water quality (and some other key ecological attributes of the water body - as discussed in section 6.10), will be met and there will be no additional Target 2 related costs over and above those of the WFD and older Directives.

With respect to air quality the situation is mixed. Current legislation and policy is delivering significant air quality improvements and as a result monitoring and models indicate that there will be a greater than 15% reduction in the overall area of ecosystems affected by acidification. Therefore the achievement of the restoration component of Target 2 will be achieved in this respect with no additional costs. But it should be noted that some sensitive ecosystems may still be impacted and therefore the maintenance component of Target 2 may not be fully met. However, current legislation will not achieve the 15% target with respect to air-borne eutrophication. Therefore, to address the challenge with respect to air-borne eutrophication it is clear that Target 2 requirements need to be considered within the analysis supporting the current review of air pollution policy being undertaken by the Commission. This would ensure that future air policy targets are aligned with these objectives and that any proposals to amend EU air policy explicitly identify if there are likely to be shortcomings in meeting the Target 2 objectives.

Lastly, it is important to bear in mind that restoration of eutrophicated and acidified ecosystems is not simply achieved by reducing pollutant loads. Although continuing damage is stopped, past damage is not necessarily restored. Therefore the restoration of previously damaged habitats will often require either every long natural rehabilitation and/or proactive habitats management, such as removal of nutrient enriched soils. The cost implications of proactive interventions are mostly captured within the ecosystem specific cost assessments (which are discussed below).

10.1.3 Practical ecosystem management, restoration and re-creation measures

This study firstly estimated the costs of achieving Target 2 **in addition to those associated with expected measures up to 2020** (ie the reference scenario) according to an even distribution of 15% restoration for each ecosystem type (ie Scenario A, as used in chapter 6), and then according to four other possible target disaggregation scenarios in chapter 7. Minimum and maximum cost estimates were also produced, which reflect the estimated minimum and maximum degradation levels (with respect to overall degradation levels and degradation resulting from specific key pressures). The estimated costs are the expenditure required in 2020, which comprise the costs of annual maintenance measures and the one-off costs of restoration and re-creation spread evenly over the 2010 to 2020 period. Costs are based on the current costs of measures and are not adjusted for anticipated inflation or other possible changes in the costs of measures in 2020.

The annual additional cost of **maintaining** the ecological condition of all ecosystems does not vary across the scenarios (as maintenance is a common basic component of Target 2) and is estimated to be between **€618 and 1,660 million**. The costs of maintaining each ecosystem vary considerably, largely due to their differing area. But costs also vary in relation to the need for management measures and the extent to which such requirements are already provided for through existing and expected measures. Hence the highest maintenance costs are for arable and forest ecosystems, requiring approximately €220 to 556 million and €196 to 488 million respectively. In contrast, there are no additional maintenance costs for lakes and rivers up to 2020 because it is anticipated that measures required under the WFD to maintain the condition of water bodies will be sufficient to achieve Target 2 maintenance needs.

The national costs of maintenance measures also varies considerably, principally in relation to the area of each Member State and the proportion of ecosystems within each that have costly maintenance requirements.

Under **Scenario A** (ie the default scenario with 15% restoration and re-creation for each ecosystem), the total average annual **restoration** costs 2020 range from **€7.79 to 10.9 billion**. Despite the even percentage restoration allocation across all ecosystems the majority of the costs would be associated with measures for arable ecosystems, as a result of their large area, high levels of degradation and the relatively high cost of restoration measures for them. **Re-creation** costs under this scenario would range from **€66 to 117 million**. The majority of the costs being for semi-natural grasslands and to a lesser extent sclerophyllous vegetation. When interpreting these results it should be borne in mind that there are no re-creation targets for arable, permanent crops or improved grasslands (as these artificial ecosystems do not merit re-creation), forests (because there has been a net increase in their area in recent decades) and mires, lakes and rivers (because they cannot normally be feasibly re-created with reasonable costs).

Scenario B attempts to reduce the cost of achieving Target 2 by focusing restoration measures on ecosystems that have the lowest unit cost for restoration (ie forests, heathland and tundra, mires, lakes and rivers (other than components covered by the WFD) and saltmarshes). This would result in a substantial reduction in the costs of **restoration**, with the total coming down to **€506 million to €1.75 billion**. In this scenario the largest proportion of the costs would be related to forest restoration measures, but there would also be significant restoration costs related to semi-natural grasslands and mires. Under this scenario, re-creation mainly focusses on heathland and tundra, dunes and beaches, and to a lesser extent sclerophyllous vegetation and inland marshes. This scenario would result in a reduction in the costs of ecosystem **re-creation**, with the total coming down to €**54 to 82 million**. In this scenario, due to its relatively large area, the greatest proportion of the costs would be related to the re-creation of sclerophyllous vegetation.

Scenario C aims to maximize biodiversity conservation benefits, by concentrating on the restoration of ecosystems that contain a high proportion of habitat types that are the focus of measures under the Habitats Directive. This would result in a focus on degraded areas of heathland and tundra, mires, dunes and beaches and saltmarsh, and little restoration of arable, permanent crops and improved grassland. Under this scenario the total cost of **restoration** would range from **€5.16 billion to 7.87 billion**. This is a substantial reduction on Scenario A, and also significantly lower than Scenarios D and E (see below), but much higher than the low cost scenario B. Despite the low percentage allocation to arable, permanent crops and improved grassland ecosystems, they still make up the predominant proportion of the costs due to their large area etc. The total additional **re-creation** cost would range from **€64 to 115 million**. Most of the cost would be associated with semi-natural grasslands and sclerophyllous vegetation due to the extensive area of these ecosystems.

Scenario D aims to maximise ecosystem services, by focusing on the restoration of ecosystems that provide the highest overall ecosystem service benefits. This would result in a relatively even spread of restoration with all ecosystems restored, though with a focus on forests, and to a lesser extent semi-natural grasslands, heathland and tundra, and mires. The costs of **restoration** under this scenario would range from **€5.87 billion to 8.59 billion**. Under Scenario D re-creation would focus on semi-natural grasslands and heathlands and tundra and to a lesser extent the inland marshes. The costs of **re-creation** under this scenario would range from **€69 to 129 million**. As under Scenario C the majority of the costs would result from the re-creation of semi-natural grasslands and sclerophyllous vegetation.

Scenario E attempts to provide a balanced set of ecosystem restoration targets that takes into account the biodiversity conservation importance of ecosystems (as assessed under Scenario C) and ecosystem services (as assessed for Scenario D). Under this scenario the percentage allocation is relatively uniform, although it focusses on heathland and tundra and mires, and to a lesser extent all other ecosystems, other than arable, permanent crops, improved grassland and sclerophyllous vegetation, which only have a 10% target. The costs of **restoration** under this scenario would range from **€6.20 billion to 8.97 billion**, with a distribution amongst ecosystem types that is very similar to that under Scenario D. **Re-creation** costs and their distribution under **Scenario E** are also similar to Scenario D, ranging from **€66 to 122 million**.

Before drawing conclusions from this analysis of costs it should be pointed out that there are many limitations that need to be borne in mind when interpreting the results. Firstly, the minimum and maximum cost estimates are often are highly divergent, sometimes differing by an order of magnitude. This is because the minimum and maximum values purely reflect the different estimates of degradation, and these are highly uncertain for many ecosystems, especially highly modified ecosystems (such as intensive farmland and plantation forests) where degradation is difficult to define and is little monitored. As a result large potential degradation ranges were often used in the study. Furthermore, many assumptions needed to be made with respect to degradation levels, that were primarily based expert opinion or extrapolation of highly incomplete data.

The estimated individual Member State costs should also be treated with particular caution as they are based on overall EU ecosystem degradation and conversion rates. Actual degradation and conversion rates will vary amongst countries, and therefore costs would be lower than projected for Member States with lower than average levels of ecosystem degradation, and higher if degradation levels are above average.

Also as a result of inadequate data, the cost estimates do not take into account a number of potentially significant costs, including:

- supporting actions such as general ecological research and monitoring, development of strategies, policies and legislation (eg to control alien invasive species), planning, development control, advice and training (eg to support the implementation of agri-environment schemes), site wardening, regulatory enforcement and awareness raising on biodiversity issues;
- species-specific measures (eg relating to particular habitat needs or translocations);
- land purchase requirements (although these probably need not be great to achieve Target 2); and
- the costs of marine ecosystem maintenance and restoration measures (although costs may be low as regulations to reduce pressures and natural recovery processes are likely to be the main means of achieving Target 2).

Furthermore, the cost estimates of meeting Target 2 are not able to take into account changes in demand and supply of ecosystem management measures, and instead assume average fixed rates (as for example obtained from agri-environment costs). As a result the costs of some actions that may be required over a high proportion of the land may be underestimated. Most importantly the costs of meeting Target 2 with regard to the maintenance of ecosystems is almost certainly significantly underestimated in this study because Target 2 requires maintenance over the entire land area. Thus very much higher payment rates than the current averages used in this study would probably be required to even approach the target, let alone achieve it. Consequently it is unrealistic to assume that the full maintenance of all areas of each ecosystem will be achieved.

A further constraint on this study has been that the maintenance cost estimates have to assume that the allocation of funds is in accordance with the theoretical need. For example payments to avert agricultural abandonment match the annual rate of abandonment. However, in reality, more wide-ranging payments covering areas at possible risk will need to be made to achieve the policy objective. But this additional required coverage cannot be estimated reliably. Therefore the estimated costs of maintenance should be treated as a theoretical minima.

Lastly it is important to emphasise that the cost estimates assume that measures included in the reference scenario will be fully implemented, and for some this will require stronger regulation and enforcement than has often occurred to date. If such regulations, such as those relating to CAP cross-compliance standards, pollution controls and measures under the WFD are not implemented as intended under the instruments then either Target 2 will not be achieved or additional funding will be required beyond that estimated here to implement alternative measures with more use of incentive measures.

In conclusion, despite all the limitations of the study, it is obvious that although many existing measures and funding instruments are contributing to the maintenance, restoration and re-creation of ecosystems in the EU, a substantial increase in funding will be required to achieve Target 2 (the potential sources of which are discussed below). In this respect the results are also consistent with the conclusions of other recent studies (as reviewed in chapter 2) although it is difficult to compare them directly. For example, the estimated costs of this study for achieving Target 2 in agricultural ecosystems appear to be substantially lower than those estimated by the Rural Land Management Costs study (Hart et al, 2011), which estimated, that the total annual costs for arable land, grassland and permanent crops would amount to some €29.2 billion. In contrast, the average total additional costs of maintenance, restoration and re-creation requirements for arable land, permanent crops, improved grasslands and semi-natural grasslands estimated in this study amount to €9.57 billion. This discrepancy almost certainly mainly reflects the narrower focus in this current study on additional costs, which therefore exclude current CAP spending on biodiversity. Also, as discussed above, the estimated costs are reduced in this study by the assumption that all measures under the reference scenario, such as cross-compliance standards, will be sufficiently implemented to fully address key pressures.

It is obvious that the additional funds needed to meet Target 2 will be substantial if heavy reliance is placed on incentive measures. This may raise the question as to whether Target 2 is achievable with this approach, and whether in practice Member States may choose to enhance the role of regulations in achieving Target 2. However, it is also important to put the estimated cost figures in context. Thus, for example the additional annual costs up to 2020 for agricultural ecosystems of €9.57 billion seem manageable when compared to the total CAP budget, which under the 2014 - 2020 MMF will amount to €363 billion. Within this the budget for the "greening" element of Pillar 1 is approximately €84 billion and in recent statements the Agriculture Commissioner has pointed to €100 billion being available for environmental purposes from the CAP over the coming programme period, including Pillar 2. Pillar 2 budgets will also receive matching national funding.

Also, as discussed in chapter 8, other EU funds such as under the ERDF (see section 8.4) have the potential to make greater contributions to the achievement of EU biodiversity targets. Although at the time of writing this report the budget allocation for these funds under the 2014-2020 MFF was uncertain, it is expected to be sufficient to contribute to restoration measures on a considerable scale.

It should also be remembered that most of the restoration and re-creation costs are one-off costs (averaged over the 2010-2020) period and therefore costs after 2020 will be much lower. Furthermore, as described below the maintenance and restoration of ecosystems will lead to significant benefits that are likely to offset many of the costs, and even provide net benefits in some cases.

10.1.4 The benefits of maintaining and restoring ecosystem and their services in accordance with Target 2

As discussed in section 2.9, there is now a wide appreciation of the economic and social values of ecosystems through the services and end benefits that they provide. Indeed this is explicitly recognised in the Biodiversity Strategy and other EU policy statements, such as the Resource Efficiency Roadmap and the Green Infrastructure Strategy. It can therefore be confidently anticipated that the maintenance and restoration of ecosystems will provide substantial benefits from the increased supply of ecosystem services that are dependent on healthy sustainably managed ecosystems (such as those related to water provisioning, carbon sequestration and storage, flood management, coastal protection, fisheries, cultural values, human health and recreation).

In Europe, it is estimated that the Natura 2000 network alone provides benefits of between €200-300 billion per annum, amounting to around 1.7 - 2.5 % of EU GDP (ten Brink et al, 2011). These benefits far exceed the estimated costs for managing the network (€5.8 billion per year) (Gantioler et al, 2010). In addition, it is conservatively estimated that the Natura 2000 network currently stores around 9.6 million tonnes of carbon, worth between €607 billion and €1,130 billion (stock value in 2010) depending on the price attached to a ton of carbon (ten Brink et al, 2011). Forest habitats contain the highest carbon value in the network (between €318 and €610 billion in 2010), followed by the dryland grassland system (106 - €197 billion in 2010) and the marine and inland water ecosystems (€92-€171 billion and €84-157 billion respectively). The increase in carbon storage benefits between 2010 and 2020 amounts to around €793 to €881 billion (lower and upper bound estimates for increase in value of carbon stock), partly due to the improved land management measures and partly due to the likely increase in the value of carbon itself. The TEEB Climate Issues Update (2009) suggested that the potential social returns after restoration can reach 79% for grasslands and 40% for woodland/shrublands when the multiple ecosystem services are taken into account.

The restoration of vulnerable ecosystems can have the additional benefit of preventing further deterioration of valuable service provision. For instance, it is estimated that 40% of Irish peatlands could be destroyed in the coming decades as a consequence of climate change (Jones et al, 2006). Although covering only 3% of the world's land surface, peatlands store more carbon than global forest biomass, constituting the world's most efficient store of carbon in terrestrial environments (Bain et al, 2011). A loss of just 1.6 % of global peat would equate to the total annual anthropogenic GHG emissions (Bain et al, 2011). Undamaged peatlands remove carbon from the atmosphere through photosynthesis of their associated vegetation and store it in the peat as plant remains. At a European scale, these prevented losses are likely to be very significant.

Water purification and provisioning are also important ecosystem services that are provided by natural ecosystems. For instance, it is estimated that about 70% of the UK's drinking water is sourced from upland areas, moorlands and heaths and these habitats buffer water quality against the effects of atmospheric, diffuse and point source pollutants (UK NEA, 2011).

Natural ecosystems can also significantly reduce the costs associated with natural disasters. The existence of forests in the Alpine region in Switzerland, for instance, is recognised as a major resource for national (public and private) socio-economic well-being in terms of disaster prevention, with 17% of Swiss forests estimated to bring value of around US $2-3.5 billion per annum in avalanche, rock fall and landslide protection (ISDR, 2004). In the UK, the benefits of salt marsh maintenance and restoration can be up to £4,600 per metre of coast owing to reduced tide and wave impact resulting in avoided costs for maintaining sea defences (Rupp and Nicholls, 2002) while Doody (2008) reports a benefit of £25 million from a UK case study compared to a restoration cost £2.61 million.

In addition to the ample evidence of the social and economic benefits of maintaining healthy ecosystems, studies have also shown that restoration benefits tend to outweigh costs, with cost-benefit ratios being in the range of 3 to 75 (Nellemann and Corcoran, 2010). But it must be borne in mind that many of the reviews have focused on ecosystems outside Europe, which have no equivalents in Europe, or which are significantly different in their ecological characteristics and processes. Therefore, it is difficult to draw conclusions on cost benefit ratios in the EU. Nevertheless, a few examples, reviewed in section 2.9.3, relating to the restoration of peatlands, rivers, coastal saltmarshes and marine ecosystems (through the establishment of marine protected areas) show that restoration in the EU can provide substantial benefits that in some case greatly outweigh costs. This highlights the need to carry out further research work to better understand and quantify ecosystem restoration benefits so that funding can be more easily sought, justified and targeted.

10.2 The potential sources of financing the maintenance and restoration of ecosystem to achieve Target 2 of the EU Biodiversity Strategy

The financing of Target 2 will require investments greater than those currently made available. Despite increased attention on private sector sources of financing, and its consequent focus in this study, the most significant contribution to the funding of Target 2 and other targets under the Biodiversity Strategy will continue to be public (national and European) funds for the foreseeable future.

10.2.1 Public financing of Target 2 measures

It is clear from this study that the CAP funds, in particular those under Pillar 2, already make major contributions to the management and restoration of ecosystems. Moreover, they have the potential to make a further vital contribution to the financing of the management and restoration measures that will be needed to achieve Target 2. In fact the proposed replacement of the four axes of rural development with six priorities, including one directly related to restoration of ecosystems, presents an opportunity to move to a flexible approach allowing the more integrated use of measures to deliver restoration priorities. However, this is only one priority and would need to be established with sufficient weight within each RDP to ensure sufficient resources are devoted to it.

A major concern is therefore the decrease in potential funding for measures that may support ecosystem management and restoration. The recently agreed cuts to the CAP budget through the MFF, which result in a greater reduction in Pillar 2 funds, will have significant implications for those rural development measures with the greatest potential to support restoration priorities, such as agri-environment-climate and forest-environment-climate measures. These cuts may be exacerbated by the proposal to allow a selection of Member States to be able to transfer funds from Pillar 2 to top up their Pillar 1 funding where the level of direct payments remains below 90% of the EU average.

Another limitation of CAP expenditure is the scope of certain elements of the policy. For example, the sums of money that are potentially to be delivered through Pillar 1 greening (in particular the EFA measure) are significant; however, they are limited in that they are only eligible on a proportion of farms, on existing agricultural land in receipt of direct payments and over a reduced area (5% as opposed to the Commission's proposed 7%). As such, the restoration of important semi-natural habitats and ecosystems outside the farmed area are not eligible and are more effectively delivered through targeted conservation actions such as those available through the Pillar 2 agri-environment-climate measure. Other measures under Pillar 2 of key importance are non-productive investments; advice, training and information measures; Payments to areas facing natural or other specific constraints, Natura measures and LEADER. In terms of the potential contribution to Target 2, environmentally focused rural development measures under Pillar 2 will continue to have the greatest impact; it is therefore essential to increase these to achieve Target 2.

10.2.2 Private sector financing of Target 2 measures

Although the most significant funding sources for biodiversity measures continue to be public (national and European) funds significant potential exists to use existing instruments in innovative ways to increase private financing. Some of the conclusions that can be drawn from this study on the factors that may influence the potential for increased private financing of conservation measures for biodiversity and related ecosystem services are therefore outlined below.

The need for the blending of public and private sector funding

Generating private finance of ecosystem restoration in the EU is a considerable challenge, and no one mechanism identified can easily provide solutions. Private sector initiatives funding biodiversity conservation can be thought of as being motivated on a scale from purely altruistic to being solely profit driven; another reason is as a result of a regulatory requirement (eg a tax or a no net loss policy). Often some combination of these factors is at work. Those that are able to pay their way and are least dependent on support from public funds are likely to be the most successful and long-lived. In addition, those for which mechanisms already exist and/or are well understood are most likely to succeed in the near future.

Private sector financing of biodiversity is already taking place, demonstrating these different motivations. Some cases are based primarily on the securing of profits in the long term (eg the Bionade payments for securing water) or the development of new markets (such as the Forest Stewardship Council). In other cases, national governments have introduced targeted environmental taxes, the proceeds of which they have invested in restoration initiatives or used to further leverage private sector funds (eg the Aggregates Levy Sustainability Fund in the UK).

However, at a European scale, these measures have yet to make a significant contribution to funding maintenance and restoration measures. Indeed, despite a substantial literature on the potential for private sector financing, remarkably few examples exist; for instance, despite apparent clear overlaps of interests, no case-studies of insurance sector involvement in risk reduction schemes relevant to Target 2 could be identified. It is also noteworthy that those schemes that are often considered to be the leading examples of profit-driven payment for ecosystem services (the funding of sustainable catchment management by Vittel (in France) and United Utilities (in the UK)), relied on the payments made under the national agri-environmental schemes in order to justify the investments made. Similar observations were made for certain product labelling and certification initiatives. Thus a combination of mechanisms using public funds to both mobilise investment and to increase returns on commercial activities are very often required in order to trigger private sector investment.

In certain respects, these findings are unsurprising. It has been widely observed that a significant reason for the loss of biodiversity is the ‘public goods’ characteristic of biodiversity resources: ie there is a public benefit that is non-exclusive and thus it is not possible for an investor to reap the full return on investment. Therefore, the use of public funds to leverage private sector investment of these schemes has a clear justification in representing the public use values provided by biodiversity.

In addition, encouraging the combination of potential private finance mechanisms themselves can result in significantly greater impact. For example, bio-carbon finance can be financed alongside other mechanisms like PES or product labelling. Potential is therefore significantly greater if this blending of funds can be expanded in a way that attracts greater private finance to ecosystem management. The rarity of such combinations is at least in part due to the increased risks, uncertainty and transactions costs of organising multi-stakeholder transactions. Public support for pilot or demonstration projects is therefore needed to progress the potential for private financing of Target 2 in this way.

Funding mechanisms available to governments to leverage private finance

Both national and European-level tools exist to leverage private sector financing. Those instruments with the greatest amounts of funds raised, as identified by the case studies, are those in which measures are introduced at the national level to either leverage funds from the private sector (eg Dutch Green Funds) or act as an environmental tax (eg UK Landfill

Levy). These increasingly offer potential in a political climate of focus on corporate social responsibility, where large businesses often have significant budgets to spend on social and environmental causes. Securing parts of these budgets for Target 2 measures will take a concerted effort from governments and the NGO sector. However, these national funding instruments require the relinquishing of tax revenue for restoration purposes or the imposing of an environmental tax, both of which may find little political support in times of austerity.

The analysis shows that the most promising instrument for the leveraging of private sector financing of measures under Target 2 is at the European level; overwhelmingly, the agri-environment measure. Its mode of operation is already well established and has a large budget capable of complementing private financing on a scale that would make a significant contribution to Target 2. By nature of its design, it also applies to the majority of the terrestrial ecosystems considered in this study and has been used successfully with private finance on numerous occasions. Therefore, reductions in the overall amount of funding available for Pillar 2 potentially represent an additional loss of potential private sector funding for the farming community and for spending on Target 2 measures.

In addition, a range of measures are available to governments, which may prove to be important in supporting private sector finance. These include:

- Shaping subsidies to better support ecosystem maintenance and restoration. Key actions in this area are to reduce the opportunity costs of restoration related to CAP pillar 1 payments, and to better channel available support to businesses who can restore ecosystems (eg through pro-biodiversity business initiatives).

- Sharing the risks of investing in ecosystem maintenance and restoration. This can use different financial instruments such as loan guarantees and offering reduced interest rates amongst others.

- Helping raise capital for up-front investment in ecosystem restoration, eg through bond issues or public private partnership arrangements, supported by the public sector through risk-reducing actions such as bond guarantees or long-term commitments to maintain funding (eg through agri-environment agreements).

- Providing incentives through the tax system for ecosystem restoration. This could be through tax relief on ecosystem restoration activities or restored ecosystems, or through hypothecation of taxes to restoration actions with a requirement for private matched-funding.

- Expanding markets for products associated with restored ecosystems, either through product labelling, or through payments for ecosystem services arrangements.

In addition the design of European funding instruments could take into greater consideration the opportunity for private sector engagement. For instance, the design of the agri-environment schemes could include flexibility to extend to areas where multiple services are provided that may allow for a combination of private sector initiatives to occur (eg payments for ecosystem services, product labelling and bio-carbon markets all on the same area of land, thus increasing the attractiveness of investment).

Given their motivations, the work of NGOs and foundations can and do make an important contribution to financing restoration projects where these appear to deliver high benefits to biodiversity but have more limited prospects as regards financial return on investment.

Establishing the right conditions required to promote private investment in nature

It is clear from the case studies that lack of money is not the only barrier to private sector investment; there also remains a gap of understanding of the opportunities and the risks related to investing in ecosystem maintenance and restoration. There can often be a gap in trust between stakeholders – private, public and external stakeholders. An important element of success stories has been the role of the third sector (NGOs and foundations) in brokering deals between these stakeholders. The capacity to broker and organise investment can also be provided by governments: pro-biodiversity business platforms are a way for governments to establish brokering and organisational capacity. This may again be most efficient when undertaken in partnership with NGOs and civil society. Therefore, an expansion of private finance for ecosystem restoration actions is more likely if there is increased capacity within NGOs and civil society to support it.

The public sector also has a role in promoting pilot projects, developing and testing the measurement and reporting of the environmental and social impacts and establishing common frameworks for action by the private sector, much in the same way as it has done for carbon markets. The establishment of dedicated policy processes to provide analysis of the current and future levels of services being supported by ecosystems and the consequences for human welfare (eg the UK Natural Capital Committee) can support decision making by investors on where the costs of maintenance and restoration are justified by the benefits.

The three most promising instruments

The following three instruments were identified as having particular potential in the near future: Payments for Ecosystem Services (PES), Product Labelling and Certification, and Bio-Carbon Markets. All directly impact on the ecosystem (ie finance changes on the ground, rather than acting as a supporting framework), are well understood - in some cases are already happening - and have potential to have large-scale influences. PES and labelling work well for maintenance measures while bio-carbon markets have the potential to support these measures where the carbon markets recognise future carbon emissions avoided. Both PES and carbon markets have a high potential for funding restoration measures. It is therefore suggested that further more detailed studies are undertaken to establish more accurately the potential for each of the funding instruments to contribute to Target 2 (and other Biodiversity Strategy targets) and to identify practical measures that may put in place to further promote and expand their use where appropriate.

10.2.3 Overall conclusions

There are a wide range of EU and national measures, including regulations and funding for practical ecosystem management and restoration that are already contributing to the objectives of Target 2. Most notably, if properly implemented, the WFD will maintain aquatic ecosystems and result in sufficient restoration to meet most aspects of Target 2 requirements. However, for all other ecosystems significant additional practical maintenance and restoration measures are clearly required to achieve Target 2.

Whilst it appears appropriate to achieve some ecosystem maintenance and restoration through new or strengthened regulations (especially regarding air pollution), most will require proactive actions encouraged through incentive payments that will therefore need significant additional funding. Some of the required funding may be obtainable from the private sector through new innovative financing mechanisms, such as PES schemes. However, it seems inevitable that up to 2020 most of the funding will need to be provided by public EU funds and national contributions. In this respect the CAP has a key role to play, especially through the adequate funding of agri-environment and similar measures that can contribute to Target 2. However, other large EU funds, especially those linked to Cohesion Policy, also have the potential to make major contributions to the achievement of the Target.

Although the costs of achieving Target 2 are significant, it is important to bear in mind the substantial benefits that would arise from the achievement of the target, and the Biodiversity Strategy as a whole. Whilst current data are insufficient to quantify exact cost-benefit ratios, there is good evidence to show that investing in the maintenance and restoration of ecosystems and their services will provide substantial environmental, social and economic gains.

11 REFERENCES

Aebischer, N J (1991) Twenty years of monitoring invertebrates and weeds in cereal fields in Sussex, in L G Fairbank, N.Carter, J F Derbyshire, & G R Potts. (eds) *The Ecology of Temperate Cereal Fields,* Blackwell Scientific Publications, Oxford.

Acuña, V., Díez, J. R., Flores, L., Meleason, M., Elosegi, A. (2013), Does it make economic sense to restore rivers for their ecosystem services?. Journal of Applied Ecology, 50: 988–997.

Airoldi, L and Beck, M W (2007) Loss, status and trend for coastal marine habitats of Europe. Gibson, RN, Atkinson, RJA, and Gordon, JDM. [45], 345-405. CRC Press. Oceanography and Marine Biology: An Annual Review.

Allen, B, Buckwell, A, Baldock, D and Menadue, H (2012a) *Maximising environmental benefits through Ecological Focus Areas*. Report commissioned by the Land Use Policy Group, Institute for European Environmental Policy, London, UK.

Allen, B, Keenleyside, C and Menadue, H (2012b) *Designed for the environment: Principles and environmental priorities for the 2014-2020 Rural Development Programmes*. Report prepared for the RSPB, Institute for European Environmental Policy, London.

Allen, H D, Randall, R E, Amable, G S and Devereux, B J (2006) The impact of changing olive cultivation practices on the ground flora of olive groves in the Messara and Psiloritis regions, Crete, Greece. *Land Degradation and Development*, No 17, (3) pp249-273.

Amann, M (2012) *Future emissions of air pollutants in Europe - Current legislation baseline and the scope for further reductions*. TSAP Report No. 1, International Institute for Applied Systems Analysis IIASA, Laxenburg, Austria.

Anderson, R (2010) *Restoring afforested peat bogs: results of current research*. 6, Forestry Commission, Roslin, UK.

Andreasen, C and Streibig, J C (2011) Evaluation of changes in weed flora in arable fields of Nordic countries - based on Danish long-term surveys. *Weed Research*, No 51, (3) pp214-226.

Armsworth, P R, Cantú-Salazar, L, Parnell, M, Davies, Z G and Stoneman, R (2011) Management costs for small protected areas and economies of scale in habitat conservation. *Biological Conservation*, No 144, pp423-429.

Aronson, J, Blignaut, J N, Milton, S J, Le Maitre, D, Esler, K J, Limouzin, A, Fontaine, C, De Wit, M P, Mugido, W, Prinsloo, P, Van Der Elst, L and Lederer, N (2010) Are socioeconomic benefits of restoration adequately quantified? A meta-analysis of recent papers (2000-2008) in Restoration Ecology and 12 other scientific journals. *Restoration Ecology*, No 18, (2) pp143-154.

Ausden, M (2007) *Habitat Management for Conservation: A Handbook of Techniques.* Oxford University Press, Oxford.

Backshall, J, Manley, J and Rebane, M (eds) (2001) *The Upland Management Handbook.* English Nature, Peterborough.

Bain, C G, Bonn, A, Stoneman, R, Chapman, S, Coupar, A, Evans, M, Gearey, B, Howat, M, Joosten, H, Keenleyside, C, Labadz, J, Lindsay, R, Littlewood, N, Lunt, P, Miller, C J, Moxey, A, Orr, H, Reed, M, Smith, P, Swales, V, Thompson, D B A, Thompson, P S, Van de Noort, R, Wilson, J D and Worrall, F (2011) *IUCN UK Commission of Inquiry on Peatlands*. IUCN UK Peatland Programme, Edinburgh.

Bal, D, Beije, H M, Fellinger, M, Haveman, R, van Opstal, A J F M and van Zadelhoff, F J (2001) *Handboek Natuurdoeltypen.* Rapport Expertisecentrum LNV 2001/020 ed. Expertisecentrum LNV, Wageningen, Netherlands.

Baldock, D, Beaufoy, G, Bennett, G and Clark, J (1993) *Nature Conservation and New Directions in the EC Common Agricultural Policy: The Potential Role of EC Policies in Maintaining Farming and Management Systems of High Nature Value in the Community*. Institute for European Environmental Policy, London.

Balmford, A, Gravestock, P, Hockley, N, McClean, C J and Roberts, C M (2004) The worldwide costs of marine protected areas. *Proceedings of the National Academy of Sciences*, No 101, (26) pp9694-9697.

Bayerisches Landesamt für Umwelt (2011) *Kostendatei für Massnahmen des Naturschutzes und der Landschaftspflege*. grundlegend überarbeitet 2010/2011. Bayerisches Landesamt für Umwelt, Augsburg, Germany.

Beaufoy, G and Marsden, K (2010) *CAP Reform 2013: last chance to stop the decline of Europe's High Nature Value farming?* Joint position paper by EFNCP, BirdLife International, Butterfly Conservation Europe and WWF. European Forum on Nature Conservation and Pastoralism, Derwentside, UK.

Beaufoy, G, Jones, G, Kazakova, Y, McGurn, P, Poux, X and Stefanova, V (2011) *Permanent Pastures and Meadows under the CAP: the situation in 6 countries*. European Forum for Nature Conservation and Pastoralism (EFNCP) and the Grasslands Trust, Derwentside.

Bennett, G and Mulongoy, K J (2006) *Review of Experience with Ecological Networks, Corridors and Buffer Zones.* 23, Secretariat of the Convention on Biological Diversity, Montreal.

Bensettiti, F and Trouvilliez, J (2009) *Rapport synthétique des résultats de la France sur l'état de conservation des habitats et des espèces conformémement à l'article 17 de la directive habitats*. Rapport SPN 2009/12, MNHN Département Ecologie et Gestion de la Biodiversité - INPN Service du Patrimoine Natural, Paris.

Berry, P, Jones, A P, Nicholls, R J and Vos, C C (2007) *Assessment of the vulnerability of terrestrial and coastal habitats and species in Europe to climate change*. Annex 2 of Planning for biodiversity as climate changes. Natural England, UK.

Billeter, R, Liira, J, Bailey, D, Bugter, R, Arens, P, Augenstein, I, Aviron, S, Baudry, J, Bukacek, R, Burel, F, Cerny, M, De Blust, G, De Cock, R, Diekotter, T, Dietz, H, Dirksen, J, Dormann, C, Durka, W, Frenzel, M, Hamersky, R, Hendrickx, F, Herzog, F, Klotz, S, Koolstra, B, Lausch, A, Le Coeur, D, Maelfait, J P, Opdam, P, Roubalova, M, Schermann, A, Schermann, N, Schmidt, T, Schweiger, O, Smulders, M J M, Speelmans, M, Simova, P, Verboom, J, van Wingerden, W K R E, Zobel, M and Edwards, P J (2008) Indicators for biodiversity in agricultural landscapes: a pan-European study. *Journal of Applied Ecology*, No 45, (1) pp141-150.

Bond, W J and Keeley, J E (2005) Fire as global 'herbivore': The ecology and evolution of flammable ecosystems. *Trends in Ecology & Evolution*, No 20, pp387-394.

Borja, A and Elliott, M (2007) What does `good ecological potential' mean, within the European Water Framework Directive? *Marine Pollution Bulletin*, No 54, (10) pp1559-1564.

Borja, A, Dauer, D, Elliott, M and Simenstad, C (2010) Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. *Estuaries and Coasts*, No 33, (6) pp1249-1260.

Bota, G, Morales, M B, Mañosa, S and Camprodon, J (eds) (2005) *Ecology and Conservation of Steppe-Land Birds.* Lynx Edicions & Centre Tecnològic Forestal de Catalunya, Barcelona, Spain.

Bragg, O, Lindsay, R and (ed.) (2003) *Strategy and Action Plan for Mire and Peatland Conservation in Central Europe*. Wetlands International, Wageningen.

Brown, J and Macfadyen, G (2007) Ghost fishing in European waters: Impacts and management responses. *Marine Policy*, No 31, (4) pp488-504.

Bruggisser, O T, Schmidt-Entling, M H and Bacher, S (2010) Effects of vineyard management on biodiversity at three trophic levels. *Biological Conservation*, No 143, (6) pp1521-1528.

Bryhn, A C (2009) Sustainable phosphorus loadings from effective and cost-effective phosphorus management around the Baltic Sea. *PLoS ONE*, No 4, (5) e5417.

Bulleri, F and Chapman, M G (2010) The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology*, No 47, (1) pp26-35.

Bullock, J M, Aronson, J, Newton, A C, Pywell, R F and Rey Benayas, J M (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution*, No 26, (10) pp541-549.

Burkhard, B, Kroll, F, Müller, F and Windhorst, W (2009) Landscapes' capacities to provide ecosystem services - a concept for land-cover based assessments. *Landscape Online*, No 15, pp1-22 DOI:10.3097/LO.200915.

Büttner, G, Feranec, G and Jaffrain, G (2006) *Corine Land Cover nomenclature illustrated guide (Addendum 2006)*. European Topic Centre on Spatial Information and Analysis (ETC/SIA).

Büttner, G, Kosztra, B, Maucha, G and Pataki, R (2012) *Implementation and achievements of CLC2006*. Revised final draft, European Topic Centre Land Use and Spatial Information (ETC LUSI), European Environment Agency.

Cadotte, M W, Carscadden, K and Mirotchnick, N (2011) Beyond species: functional diversity and the maintenance of ecological processes and services. *Journal of Applied Ecology*, No 48, (5) pp1079-1087.

Calaciura, B and Spinelli, O (2008a) *Management of Natura 2000 Habitats: 6210 *Semi-natural dry grasslands and scrubland facies on calcareous substrates (Festuco-Brometalia)*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 12/24, European Commission, Brussels.

Calaciura, B and Spinelli, O (2008b) *Management of Natura 2000 Habitats: 5210 Arborescent matorral with Juniperus spp.* Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 10/24, European Commission, Brussels.

Cao, Y, Elliott, J, McCracken, D, Rowe, K, Whitehead, J and Wilson, L (2009) *Estimating the Scale of Future Environmental Land Management Requirements for the UK*. Land Use Policy Group.

Cardenas, L M, Cuttle, S P, Crabtree, B, Hopkins, A, Shepherd, A, Scholefield, D and del Prado, A (2011) Cost effectiveness of nitrate leaching mitigation measures for grassland livestock systems at locations in England and Wales. *Science of the Total Environment*, No 409, (6) pp1104-1115.

Cardenas, M, Ruano, F, Garcia, P, Pascual, F and Campos, M (2006) Impact of agricultural management on spider populations in the canopy of olive trees. *Biological Control*, No 38, (2) pp188-195.

CBD (2011) Quick Guides for the Aichi Biodiversity Targets. Convention on Biological Diversity https://www.cbd.int/nbsap/training/quick-guides/

CEC (2006) *Communication of the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions - Thematic Strategy for Soil Protection*. COM(2006)231 final, 22.9.2006, Commission of the European Communities, Brussels.

Cichanowicz, J E (2010) *Current capital cost and cost-effectiveness of power plant emissions control technologies*. Paper for the Utility Air Regulatory Group. American Public Power Association.

Conley, D J, Carstensen, J, Aigars, J, Axe, P, Bonsdorff, E, Eremina, T, Haahti, B-M, Humborg, C, Jonsson, P, Kotta, J, Lännegren, C, Larsson, U, Maximov, A, Medina, M R, Lysiak-Pastuszak, E, Remeikaite-Nikiene, N, Walve, J, Wilhelms, S and Zillén, L (2011) Hypoxia is increasing in the coastal zone of the Baltic Sea. *Environmental Science and Technology*, No 45, (16) pp6777-6783.

Conrad, M K and Tischew, S (2011) Grassland restoration in practice: Do we achieve the targets? A case study from Saxony-Anhalt/Germany. *Ecological Engineering*, No 37, (8) pp1149-1157.

Conrady, D, Johst, A, Kathke, S, Rösch, W, Baasch, A, Runge, K, Grosser, N, Krebs, V, Findeisen, E and Hering, T (2012) *Energieholz und Biodiversität - Die Nutzung von Energieholz als Ansatz zur Erhaltung und Entwicklung national bedeutsamer Lebensräume*. 3. Zwischenbericht des Projektes. Unveröffentlichter Bericht.

Cots-Folch, R, Martínez-Casasnovas, J A and Ramos, M C (2006) Land terracing for new vineyard plantations in the north-eastern Spanish Mediterranean region: Landscape effects of the EU Council Regulation policy for vineyards' restructuring. *Agriculture, Ecosystems & Environment*, No 115, (1-4) pp88-96.

COWI (2010) *Compliance Costs of the Urban Wastewater Treatment Directive*. Final report for DG ENV, European Commission DG Environment, Brussels.

Cranford, M, Parker, C and Trivedi, M (2011) *Understanding forest bonds: a guide to raising up-front finance for tropical forests*. Global Canopy Programme, Oxford, UK.

Cullis-Suzuki, S and Pauly, D (2010) Marine protected area costs as 'beneficial' fisheries subsidies: a global evaluation. *Coastal Management*, No 38, (2) pp113-121.

Davy, C M, Russo, D and Fenton, M B (2007) Use of native woodlands and traditional olive groves by foraging bats on a Mediterranean island: consequences for conservation. *Journal of Zoology*, No 273, (4) pp397-407.

de Graaff, J, Duarte, F, Fleskens, L and de Figueiredo, T (2010) The future of olive groves on sloping land and ex-ante assessment of cross compliance for erosion control. *Land Use Policy*, No 27, (1) pp33-41.

de Groot, R S (2008) Socio-economic benefits of ecosystem restoration and nature conservation: empirical evidence that investing in natural capital pays, in *6th European Conference on Ecological Restoration,* 8.9.2008, Ghent, Belgium

DEFRA (2002) *Coastal realignment review project report*. Policy Research Project FD 2008, Defra / Environment Agency, London.

DEFRA (2006) *National evaluation of the costs of meeting coastal environmental requirements*. R&D Technical Report FD2017/TR, Defra.

DEFRA (2007) *An economic analysis to inform the Air Quality Strategy*. Updated third report of the interdepartmental group on costs and benefits. DEFRA.

Department of Agriculture (2012) Rural Development Programme. Nicosia, Cyprus, Republic of Cyprus.

Dickie, I A, Gantioler, S, Cooper, G, McConville, A J, Dutton, A, Burrett, R, Duke, G and Treweek, J R (2012) *Innovative use of financial instruments and approaches to enhance* *private sector finance of biodiversity*. 1070307/2010/581922/etu/f1, Final Summary Report to the European Commission Directorate-General Environment.

Ding, W, Zhang, Y and Cai, Z (2010) Impact of permanent inundation on methane emissions from a *Spartina alterniflora* coastal salt marsh. *Atmospheric Environment*, No 44, (32) pp3894-3900.

Dise, N B (2011) Nitrogen as a threat to European terrestrial diversity, in M A Sutton, C M Howard, J W Erisman, G Billen, A Bleeker, P Grennfelt, H van Grinsven, & B Grizzetti (eds) *The European Nitrogen Assessment,* ppChapter 20. Cambridge University Press, Cambridge.

Doherty, J M, Callaway, J C and Zedler, J B (2011) Diversity–function relationships changed in a long-term restoration experiment. *Ecological Applications*, No 21, (6) pp2143-2155.

Donald, P F (1998) Changes in the abundance of invertebrates and plants on British farmland. *British wildlife*, No 9, pp279-289.

Doody, J P (2008) *Management of Natura 2000 Habitats: 1330 Atlantic salt meadows (Glauco-Puccinellietalia maritimae)*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 02/24, European Commission, Brussels.

Duarte, F, Jones, N and Fleskens, L (2008) Traditional olive orchards on sloping land: Sustainability or abandonment? *Journal of Environmental Management*, No 89, (2) pp86-98.

Dworak, T, Berglund, M, Thaler, T, Fabik, E, Amand, B, Grandmougin, B, Ribeiro, M M, Laaser, C and Matauschek, M (2010) *Assessment of agriculture measures included in the draft River Basin Management Plans - Summary Report*. This report is a deliverable for the project ENV.D.2/SER/2009/0002r, Ecologic, Berlin/Vienna.

EASAC (2009) *Ecosystem Services and Biodiversity in Europe*. EASAC Policy Report 09, European Academies Science Advisory Council Secretariat, The Royal Society, London, UK.

EC (2010) *Green Paper on Forest Protection and Information in the EU: Preparing forests for climate change*. COM (2010)66, 1.3.2010, European Commission, Brussels.

Ecorys (2011) *actualisatie kostenraming maatregelen WMIJ (update cost projections ijsselmeer restoration)*.

EEA (2005) *Source apportionment of nitrogen and phosphorus inputs into the aquatic environment*. EEA Report No 7/2005, European Environment Agency, Copenhagen.

EEA (2006a) *European forest types: Categories and types for sustainable forest management reporting and policy*. No 9/2006, European Environment Agency, Copenhagen.

EEA (2006b) *The changing faces of Europe's coastal areas*. EEA Report No 6/2006, European Environment Agency, Copenhagen.

EEA (2007) *Europe's Environment: The Fourth Assessment*. State of the Environment Report No.1/2007, European Environment Agency, Copenhagen.

EEA (2008) *European forests - ecosystem conditions and sustainable use*. EEA Report No 3/2008, European Environment Agency, Copenhagen.

EEA (2009a) *Progress Towards the European 2010 Biodiversity Target - Indicator Fact Sheets*. The European Environment Agency, Copenhagen.

EEA (2009b) *Water Resources across Europe. Confronting Water Scarcity and Drought*. EEA Report No 2/2009, European Environment Agency, Copenhagen.

EEA (2009c) *Regional climate change and adaptation - The Alps facing the challenge of changing water resources*. EEA Report No 8/2009, European Environment Agency, Copenhagen.

EEA (2010a) *EU 2010 Biodiversity Baseline*. EEA Technical Report No 12/2010, European Environment Agency, Copenhagen.

EEA (2010b) *Assessing Biodiversity in Europe - the 2010 Report*. EEA Technical Report No 5/2010, European Environment Agency, Copenhagen.

EEA (2010c) The European Environment State and Outlook 2010: Synthesis. 1-228. Copenhagen, European Environment Agency. The European Environment State and Outlook SOER.

EEA (2010d) *CSI Indicators. Nutrients in freshwater (CSI 020)*. Assessment published Dec 2010. EEA Report, European Environment Agency, Copenhagen.

EEA (2010e) The European Environment State and Outlook 2010: Landuse. Copenhagen, European Environment Agency. The European Environment State and Outlook SOER.

EEA (2010f) *Ecosystem accounting and the cost of biodiversity losses. The case of coastal Mediterranean wetlands*. EEA Report No 3/2010, European Environment Agency, Copenhagen.

EEA (2011) *CSI Indicators. Nutrients in transitional, coastal and marine waters (CSI 021)*. Assessment published Jul 2011. European Environment Agency, Copenhagen.

EEA (2012a) *CSI Indicators. Exposure of ecosystems to acidification, eutrophication and ozone (CSI 005)*. Assessment published November 2012. European Environment Agency.

EEA (2012b) *The impacts of invasive alien species in Europe*. EEA Technical report No16/2012, European Environment Agency, Copenhagen.

EEA (2012c) *Environmental Indicator Report 2012*. Ecosystem resilience and resource efficiency in a green economy in Europe. European Environment Agency, Copenhagen.

EEA (2012d) *European waters - assessment of status and pressures*. EEA Report No.8/2012, European Environment Agency, Copenhagen.

Elliott, M, Burdon, D, Hemingway, K L and Apitz, S E (2007) Estuarine, coastal and marine ecosystem restoration: Confusing management and science- A revision of concepts. *Estuarine, Coastal and Shelf Science*, No 74, (3) pp349-366.

English Nature (2005) *The Importance of Livestock Grazing for Wildlife Conservation.* English Nature, Peterborough.

Environment Agency (2010) *Working with natural processes to manage flood and coastal erosion risk*. A Guidance Document, Environment Agency, Bristol.

Environment Agency (2011) *The extent of saltmarsh in England and Wales: 2006-2009*. Environment Agency, Bristol.

Eriksson, M O G (2008) *Management of Natura 2000 Habitats: 6450 Northern Boreal alluvial meadows*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 18/24, European Commission, Brussels.

Ernst & Young (2011) *Interim Evaluation of the European Fisheries Fund (2007 - 2013)*. Final report, European Commission, Brussels.

ETC/BD (2006) *Assessment, monitoring and reporting under Article 17 of the Habitats Directive: Explanatory Notes & Guidelines*. Final Draft. European Topic Centre on Biological Diversity, Paris.

ETC/BD (2008a) *Habitats Directive Article 17 Technical Report (2001-2006)*. European Topic Centre on Biological Diversity, Paris.

ETC/BD (2008b) *Habitats Directive Article 17 Technical Report (2001-2006): Overview of conservation status*. European Topic Centre on Biological Diversity, Paris.

ETC/ICM (2012a) *Hydromorphological alterations and pressures in European rivers, lakes, transitional and coastal waters*. Thematic assessment for EEA Water 2012 Report. EEA Topic Centre on Inland, Coastal and Marine Waters.

ETC/ICM (2012b) *Ecological and chemical status and pressures in European waters*. Thematic assessment for EEA Water 2012 Report. ETC/ICM Technical Report 1/2012, EEA Topic Centre on Inland, Coastal and Marine Waters.

EUC Coastal & Marine Union (2010) *Carbon Sequestration & Biodiversity*. EAF supported implementation of biodiversity objective that can contribute to emission reduction aims. Final Report of the project EUCC's contribution to EU Marine Policy and EU Climate Policy / Carbon Sequestration and Biodiversity, EUC Coastal & Marine Union and EEECONET ACTION FUND, http://www.eeconet.org/eaf/climate-biodiversity/cbmain.html.

European Commission (2003) *Natura 2000 and Forests, Challenges and Opportunities. Interpretation Guide*. European Commission, Brussels.

European Commission (2007) *LIFE and Europe's Wetlands*. Restoring a vital ecosystem. Office for Official Publications of the European Communities, Luxembourg.

European Commission (2009) *Commission Staff Working Document. Peak Performance. New insights into mountain farming in the European Union*. SEC (2009) 1724 FINAL, 16.12.2009, European Commission, Brussels.

European Commission (2010) *LIFE among the olives: good practice in improving environmental performance in the olive oil sector*. Publications Office of the European Union, Luxembourg.

European Commission (2011a) *Our life insurance, our natural capital: an EU biodiversity strategy to 2020*. COM(2011)244 Final, 3.5.2011a, European Commission, http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/1_EN_ACT_part1_v7%5b1%5d.pdf.

European Commission (2011b) *Commission Staff Working Paper. Impact Assessment. Accompanying the document Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: Our life insurance, our natural capital: an EU biodiversity strategy to 2020*. SEC(2011) 540 final, 3.5.2011b, European Commission, Brussels.

European Commission (2011c) *Guidelines on the implementation of the Birds and Habitats Directives in estuaries and coastal zones*. Guidance Document.

European Commission (2012) Prospects for the olive oil sector in Spain, Italy and Greece - 2012-2020. [Brief No.2 - July 2012]. DG Agriculture & Rural Development, European Commission. Agricultural Markets Briefs.

European Commission (2013) *Green Infrastructire (GI) – enhancing Europe's natural capital*. COM(2013) 249 final. 6.5.2013, European Commission, Brussels.

European Court of Auditors (2011) *Have EU measures contributed to adapting the capacity of the fishing fleets to available fishing opportunities?* Special Report No 12, Luxembourg.

EUROSION (2004a) *Living with coastal erosion in Europe: Sediment and space for sustainability (PART 2)*. PART 2 - Maps and Statistics (EUROSION project). European Commission.

EUROSION (2004b) *A guide to coastal erosion management practices in Europe: lessons learned*. EUROSION project. Service contract B4-3301/2001/329175/MAR/B3 "Coastal erosion - Evaluation of the need for action" for DG Environment, EUROSION.

Farmer, M, Cooper, T, Baldock, D, Tucker, G, Eaton, R, Hart, K, Bartley, J, Rayment, M, Arblaster, K, Beaufoy, G, Pointereau, P, Coulon, F, Herodes, M, Kristensen, L, Andersen, E, Landgrebe, R, Naumann, S, Povellato, A, Trisorio, A, Jongman, R and Bunce, B (2008) *Final Report - Reflecting Environmental Land Use Needs into EU Policy: Preserving and Enhancing the Environmental Benefits of Unfarmed Features on EU Farmland*. Report prepared for DG Environment, Contract No. ENV.B.1/ETU/2007/0033, Institute for European Environmental Policy, London.

Farrell, C A (2008) *Restoration of peatlands in Ireland*. Pole-relais Tourbieres, France.

Feilen, A (2006) *Marine Protected Areas*. SPICe Briefing 06/110, Scottish Parliament Information Centre, Edinburgh.

Fischer, R, Waldner, P, Carnicer, J, Coll, M, Dobbertin, M, Ferretti, M, Hansen, K, Kindermann, G, Lasch-Born, P, Lorenz, M, Marchetto, A, Meining, S, Nieminen, T, Peñuelas, J, Rautio, P, Reyer, C, Roskams, P and Sánchez, G (2012) *The Condition of Forests in Europe*. ICP Executive Report 2012. [http://icp-forests.net/page/icp-forests-executive-report], ICP Forests and European Commission, Hamburg and Brussels.

Forbes, B C and Kumpula, T (2009) The ecological role and geography of reindeer (*Rangifer tarandus*) in Northern Eurasia. *Geography Compass*, No 3, (4) pp1356-1380.

Forbes, B C, Bölter, M, Müller-Wille, L, Hukkinen, J, Müller, F, Gunslay, N and Konstantinov, Y (2006) Reindeer management in northernmost Europe: linking practical and scientific knowledge in social-ecological systems. Ecological Studies [184]. Berlin, Germany, Springer-Verlag.

Forest Europe, UNECE and FAO (2011) *State of Europe's Forests 2011*. Status and Trends in Sustainable Forest Management in Europe. Forest Europe, Madrid.

Forestry Commission (2000) *The impact of deer on woodland biodiversity*. Forestry Commission, Edinburgh.

Förster, J and Schäfer, A (2010) *TEEBcase: Peatland restoration for carbon sequestration, Germany*. European Environment Agency, Copenhagen.

Freistaat Thüringen (2003) *Die Eingriffsregelung in Thüringen*. Kostendateien für Ersatzmassnahmen im Rahmen der naturschutzrechtlichen Eingriffsregelung [standard costs for conservation measures]. Thüringer Ministerium für Landwirtschaft, Naturschutz und Umwelt, Germany.

Friends of the Irish Environment (2011) *The Destruction of Ireland's Protected Raised Bogs*. A report by Friends of the Irish Environment on the continuing cutting of Ireland's Natura 2000 raised bogs. Friends of the Irish Environment, Eyeries.

Frost, H, Boom, J-T, Buisman, E, Innes, J, Metz, S, Rodgers, P and Taal, K (2007) *NECESSITY Economic Impact Assessment of Changes in Fishing Gear*. Report no. 194, Institute of Food and Resource Economics, Copenhagen.

Fukami, T, Dickie, I A, Paula Wilkie, J, Paulus, B C, Park, D, Roberts, A, Buchanan, P K and Allen, R B (2010) Assembly history dictates ecosystem functioning: evidence from wood decomposer communities. *Ecology Letters*, No 13, (6) pp675-684.

Gaisler, J, Pavlu, V and Pavlu, L (2011) Effect of different extensive management treatments on the plant diversity of upland meadow without forage utilisation, in E M Pötsch, B Krautzer, & A Hopkins (eds) *Grassland Farming and Land Management Systems in Mountainous Regions,* pp577-579. Proceedings of the 16th Symposium of the European Grassland Federation Gumpenstein, Austria August 29th - August 31st 2011. European Grassland Federation, Galvánek, D and Janák, M (2008) *Management of Natura 2000 Habitats: 6230 Species-rich Nardus grasslands*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 14/24, European Commission, Brussels.

Gantioler, S, Rayment, M, Bassi, S, Kettunen, M, McConville, A J, Landgrebe, R, Gerdes, H and ten Brink, P (2010) *Costs and Socio-Economic Benefits associated with the Natura 2000 Network*. Final Report to the European Commission, DG Environment on Contract ENV.B.2/SER/2008/0038, Institute for European Environmental Policy / GHK / Ecologic, Brussels.

García, R R, Fraser, M D, Celaya, R, Ferreira, L M M, García, U and Osoro, K (2012) Grazing land management and biodiversity in the Atlantic European heathlands: a review. *Agroforestry Systems*, No 87, (1) pp19-43.

GHK (2006) *UK Biodiversity Action Plan: Preparing Costings for Species and Habitat Action Plans*. Costings Summary Report Revised Report to Defra and Partners. GHK, London.

Gilbert, O L and Anderson, P (1998) *Habitat creation and repair*. Oxford University Press.

Gimingham, C H (1992) *The Lowland Heathland Management Handbook*. 8, English Nature Science, English Nature, Peterborough.

Gimingham, C H, Chapman, S B and Webb, N R (1979) European heathlands, in R L Specht (ed) *Heathlands and Related Shrublands. A. Descriptive Studies,* Elsevier, Amsterdam.

Gobin, A, Campling, P, Janssen, L, Desmet, N, van Delden, H, Hurkens, J, Lavelle, P and Berman, S (2011) *Soil organic matter management across the EU - best practices, constraints and trade-offs*. Final Report for the European Commission, DG Environment. Technical Report - 2011 - 051, European Communities, Luxembourg.

Gómez, J A, Llewellyn, C, Basch, G, Sutton, P B, Dyson, J S and Jones, C A (2011) The effects of cover crops and conventional tillage on soil and runoff loss in vineyards and olive groves in several Mediterranean countries. *Soil Use Management*, No 27, (4) pp502-514.

González del Tánago, M, García de Jalón, D and Román, M (2012) River restoration in Spain: theoretical and practical approach in the context of the European Water Framework Directive. *Environmental Management*, No 50, (1) pp123-139.

Gravestock, P, Roberts, C M and Bailey, A (2008) The income requirements of marine protected areas. *Ocean & Coastal Management*, No 51, (3) pp272-283.

Gren, I M (1995) Costs and benefits of restoring wetlands. Two Swedish case studies. *Ecological Engineering*, No 4, (2) pp153-162.

Gren, I-M, Jonzon, Y and Lindqvist, M (2008) *Costs of nutrient reductions to the Baltic Sea - technical report*. Working Paper Series 2008:1, Swedish University of Agricultural Sciences, Uppsala.

Grootjans, A P, Geelen, H W T, Jansen, A J M and Lammerts, E J (2002) Restoration of coastal dune slacks in the Netherlands. *Hydrobiologia*, No 478, (1-3) pp181-203.

Guimaraes, A and Olmeda, C (2008) *Management of Natura 2000 Habitats: 9360 Macronesian laurel forests (Laurus, Ocotea)*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 23/24, European Commission, Spain.

Güthler, W and Oppermann, R (2005) *Agrarumweltprogramme und Vertragsnaturschutz weiter entwickeln*. Mit der Landwirtschaft zu mehr Natur: Ergebnisse des F+E-Projektes. Heft 13, Bundesamt für Naturschutz, Bonn - Bad Godesberg.

Gutman, P (2003) *From good will to payments for environmental services: a survey of financing options for sustainable natural resource management in developing countries*. WWF/MPO, Washington DC.

Halada, L, Evans, D, Romão, C and Petersen, J-E (2011) Which habitats of European importance depend on agricultural practices? *Biodiversity and Conservation*, No 20, (11) pp2365-2378.

Halassy, M and Török, K (2004) Combination of treatments restores black locust plantations to native sandy grassland (Hungary). *Ecological Restoration*, No 22, (3) pp217-218.

Hampicke, U (2010) *Expert Report on the Level of Compensation Payments for the Near-Natural Exploitation of Agricultural Land in Germany*.

Hampton, M (2008) *Management of Natura 2000 Habitats: 4010 Northern Atlantic wet heaths with Erica tetralix*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. European Commission, Brussels.

Hansen, H O (1996) *River Restoration - Danish Experience and Examples.* National Environmental Research Institute, Ministry of Environment and Energy, Denmark.

Hart, K and Little, J (2012) Environmental approach of the CAP legislative proposal. *International Agricultural Policy*, No 1, pp19-30.

Hart, K and Menadue, H (2013) Political agreement reached for the CAP reform package. IEEP Analysis briefing for the European Intelligence Service. 28 June 2013. http://www.eis.ieep.eu/land-use-management-freshwater/2013/06/political-agreement-reached-for

Hart, K, Allen, B, Lindner, M, Keenleyside, C, Burgess, P J, Eggers, J and Buckwell, A (2013) *Land as an environmental resource*. Report prepared for DG Environment, Contract No ENV.B.1/ETU/2011/0029, Institute for European Environmental Policy, London.

Hart, K, Baldock, D, Tucker, G M, Allen, B, Calatrava, J, Black, H, Newman, S, Baulcomb, C, McCracken, D and Gantioler, S (2011) *Costing the Environmental Needs Related to Rural Land Management*. Report prepared for DG Environment, Contract No ENV.F.1/ETU/2010/0019r, Institute for European Environmental Policy, London.

Hendrickx, F, Maelfait, J P, van Wingerden, W, Schweiger, O, Speelmans, M, Aviron, S, Augenstein, I, Billeter, R, Bailey, D, Bukacek, R, Burel, F, Diekotter, T, Dirksen, J, Herzog, F, Roubalova, M, Vandomme, V and Bugter, R (2007) How landscape structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes. *Journal of Applied Ecology*, No 44, (2) pp340-351.

Herremans, J-P, Houston, J, Jepsen, B, Racinska, I, Sliva, J, Rossi, I, Sopeña, A, Valaoras, G and Weeger, C (2013) *Study on the costs linked to habitat restoration within LIFE Nature projects*. Study and annexes (rivers and grasslands). EHT4 Astrale D contract, Astrale LIFE team.

Hjerp, P, Medarova-Bergstrom, K, Skinner, I, ten Brink, P, Medhurst, J, Hausemer, P, Peterlongo, G, Kalinka, P, Kettunen, M, Cachia, H, Grubbe, M and Evers, D (2011) *Cohesion Policy and Sustainable Development. A report for DG Regio.* Final Report. Institute for European Environmental Policy, London.

HM Government (2011) *The Natural Choice: securing the value of nature*. UK Natural Environment White Paper. HM Government, UK.

Hodgson, J A, Thomas, C D, Wintle, B A and Moilanen, A (2009) Climate change, connectivity and conservation decision making: back to basics. *Journal of Applied Ecology*, No 46, pp964-969.

Hoffmann, C C and Baattrup-Pedersen, A (2007) Re-establishing freshwater wetlands in Denmark. *Ecological Engineering*, No 30, (2) pp157-166.

Holden, J, Chapman, P J and Labadz, J C (2004) Artificial drainage of peatlands: hydrological and hydrochemical processes and wetland restoration. *Progress in Physical Geography*, No 28, (1) pp95-123.

House of Lords (2012) *An Indispensable Resource: EU Freshwater Policy*. 33rd Report of Session 2010-12. HL 296.

Houston, J A (2008a) *Management of Natura 2000 Habitats: 2190 Humid dune slacks*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 05/24, European Commission, Brussels.

Houston, J A (2008b) *Management of Natura 2000 Habitats: 2130 Fixed coastal dunes with herbaceous vegetation ("grey dunes")*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 04/24, European Commission, Brussels.

Howe, M A, Knight, G T and Clee, C (2010) The importance of coastal sand dunes for terrestrial invertebrates in Wales and the UK, with particular reference to aculeate Hymenoptera (bees, wasps & ants). *Journal of Coastal Conservation*, No 14, (2) pp91-102.

Hulea, O, Ebert, S and Strobel, D (2009) Floodplain restoration along the Lower Danube: A climate change adaptation case study. *IOP Science*, No 6, (402002) 1.

ICPDR (2012) International Commission for the Protection of the Danube River (Website). http://www.icpdr.org/main/

IEEP, GHK and TEPR (2012) *Background Study Towards Biodiversity Proofing of the EU Budget*. Report to the European Commission, Institute for European Environmental Policy, London.

ISDR (2004) *Living with Risk: A global review of disaster reduction initiatives*. International Strategy for Risk Disaster, UN/ISDR, Geneva, Switzerland.

Jacobsen, B H (2004) *Økonomisk slutevaluering af Vandmiljøplan II. [Final economic evaluation of Action Plan II for the aquatic environment].* Report No. 169, Institute of Food and Resource Economics, Copenhagen.

Jeffery, S, Gardi, C, Jones, A, Montanarella, L, Marmo, L, Miko, L, Ritz, K, Peres, G, Römbke, J and Van der Putten, W H (2010) *European Atlas of Soil Biodiversity.* 1st edition ed. Publications Office of the European Union, Luxembourg.

Jiang, L, Brady, L and Tan, J (2011) Species diversity, invasion and alternative community sites in sequentially assembled communities. *The American Naturalist*, No 178, (3) pp411-418.

JNCC (2007) *Second Report by the UK under Article 17 on the implementation of the Habitats Directive from January 2001 to December 2006*. Joint Nature Conservation Committee, Peterborough, UK.

JNCC (2011) *Towards an Assessment of the State of UK Peatlands*. JNCC Report No. 445, Joint Nature Conservation Committee, Peterborough.

JNCC (2012) UK Biodiversity Action Plan Priority Species. http://jncc.defra.gov.uk/page-5717

Jones, A, Panagos, P, Barcelo, S, Bouraoui, F, Bosco, C, Dewitte, O, Gardi, C, Erhard, M, Hervás, J, Hiederer, R, Jeffery, S, Lükewille, A, Marno, L, Montanarella, L, Olazábal, C, Petersen, J-E, Penizek, V, Strassburger, T, Tóth, G, Van Den Eeckhaut, M, Van Liedekerke, M, Verheijen, F, Viestova, E and Yigini, Y (2012) *The State of Soil in Europe*. A contribution of the JRC to the European Environment Agency's Environment State and Outlook Report - SOER 2010. Publications Office of the European Union, Luxembourg.

Jones, M, Donnelly, A and Albanito, F (2006) Responses of Irish Vegetation to Future Climate Change. *Biology & Environment: Proceedings of the Royal Irish Academy*, No 106, (3) pp323-334.

Jones, R J A, Hiederer, R, Rusco, E, Loveland, P J and Montanarella, L (2004) *The map of organic carbon content in topsoils in Europe. Version 1.2 September - 2003 (S.P.I.04.72)*. (explanation of Special Publication Ispra 2004 No.72 and 1 map in ISO B1 format). European Soil Bureau Research Report No.15, EUR 21209 EN, Office for Official Publications of the European Communities, Luxembourg.

Jongman, R H G and Pungetti, G (eds) (2004) *Ecological Networks and Greenways. Concept, Design, Implementation.* Cambridge University Press, Cambridge, UK.

Joosten, H (2009) *The global peatland CO2 picture. Peatland status and drainage related emissions in all countries of the world*. Wetlands International.

José-María, L, Armengot, L, Blanco-Moreno, J M, Bassa, M and Sans, F X (2010) Effects of agricultural intensification on plant diversity in Mediterranean dryland cereal fields. *Journal of Applied Ecology*, No 47, (4) pp832-840.

JRC (2012) Annual Reports on Forest Fires in Europe. JRC EFFIS - European Forest Fire Information System http://effis.jrc.ec.europa.eu/reports/fire-reports

Kampa, E and Laaser, C (2009) *Heavily Modified Water Bodies: Information Exchange on Designation, Assessment of Ecological Potential, Objective Setting and Measures*. Discussion Paper for Common Implementation Strategy Workshop Brussels, 12-13 March 2009. Ecologic Institute, unpublished.

Kaphengst, T, Bassi, S, Davis, M, Gardner, S, Herbert, S, Lago, M, Naumann, S, Pieterse, M and Rayment, M (2010) *Taking into account opportunity costs when assessing costs of biodiversity and ecosystem action*. A report for DG Environment, Ecologic, IEEP, GHK, Berlin.

Keeley, J E, Bond, W J, Bradstock, R A, Pausas, J G and Rundel, P W (2012) *Fire in Mediterranean Ecosystems: Ecology, Evolution and Management.* Cambridge University Press., Cambridge, UK.

Keenleyside, C and Tucker, G M (2010) *Farmland Abandonment in the EU: an Assessment of Trends and Prospects*. Report for WWF. Institute for European Environmental Policy, London.

Keenleyside, C, Allen, B, Hart, K, Menadue, H, Stefanova, V, Prazan, J, Herzon, I, Clement, T, Povellato, A, Maciejczak, M and Boatman, N D (2012) *Delivering environmental benefits through entry-level agri-environment schemes in the EU*. Report Prepared for DG Environment, Project ENV.B.1/ETU/2010/0035. Institute for European Environmental Policy, London.

Kettunen, M, McConville, A J and van Vliet, W (2012) *Financing biodiversity in the context of the European Fund for Regional Development (EFRD): Practical guidance based on the lessons learned from SURF Nature project (ERDF Interreg IVC)*. Published by SURF-Nature Project.

Kettunen, M, Terry, A, Tucker, G M and Jones, A (2007) *Guidance on the maintenance of landscape connectivity features of major importance for wild flora and fauna. Guidance on the implementation of Article 3 of the Birds Directive (79/409/EEC) and Article 10 of the Habitats Directive (92/43/EEC)*. Institute for European Environmental Policy, Brussels / London.

Kettunen, M, Genovesi, P, Gollasch, S, Pagad, S, Starfinger, U, ten Brink, P and Shine, C (2009) *Technical support to EU strategy on invasive alien species (IAS) - Assessment of the* *impacts of IAS in Europe and the EU*. (final module report for the European Commission). Service contract No 070307/2007/483544/MAR/B2, Institute for European Environmental Policy (IEEP), Brussels, Belgium.

Kettunen, M, Baldock, D, Gantioler, S, Carter, O, Torkler, P, Arroyo Schnell, A, Baumueller, A, Gerritsen, E, Rayment, M, Daly, E and Pieterse, M (2011) *Assessment of the Natura 2000 co-financing arrangements of the EU financing instrument*. A project for the European Commission - final report. 138 pp. + Annexes, Institute for European Environmental Policy, Brussels, Belgium.

King, M (2010) *An investigation into policies affecting Europe's semi-natural grasslands*. A report by the Grasslands Trust. European Forum on Nature Conservation and Pastoralism; European Commission (DG Environment), Derwentside, UK.

Klimkowska, A, Dzierza, P, Brzezinska, K, Kotowski, W and Medrzycki, P (2010a) Can we balance the high costs of nature restoration with the method of topsoil removal? Case study from Poland. *Journal for Nature Conservation*, No 18, (3) pp202-205.

Klimkowska, A, Kotowski, W, van Diggelen, R, Grootjans, A P, Dzierza, P and Brzezinska, K (2010b) Vegetation re-development after fen meadow restoration by topsoil removal and hay transfer. *Restoration Ecology*, No 18, (6) pp924-933.

Koehler, A K, Sottocornola, M and Kiely, G (2011a) How strong is the current carbon sequestration of an Atlantic blanket bog? *Global Change Biology*, No 17, (1) pp309-319.

Koehler, A K, Sottocornola, M and Kiely, G (2011b) How strong is the current carbon sequestration of an Atlantic blanket bog? *Global Change Biology*, No 17, (1) pp309-319.

Kovács-Hostyánszki, A, Batáry, P and Báldi, A (2011a) Local and landscape effects on bee communities of Hungarian winter cereal fields. *Agricultural and Forest Entomology*, No 13, (1) pp59-66.

Kovács-Hostyánszki, A, Batáry, P, Peach, W J and Báldi, A (2011b) Effects of fertilizer application on summer usage of cereal fields by farmland birds in central Hungary. *Bird Study*, No 58, (3) pp330-337.

Kronbak, L G, Nielsen, J R, Jorgensen, O A and Vestergaard, N (2009) Bio-economic evaluation of implementing trawl fishing gear with different selectivity. *Journal of Environmental Management*, No 90, (11) pp3665-3674.

Kuhlman, T, Reinhard, S and Gaaff, A (2010) Estimating the costs and benefits of soil conservation in Europe. *Land Use Policy*, No 27, (1) pp22-32.

Kuik, O (2006) *Ex ante and ex post costs of implementing the Nitrates Directive*. Case study in the framework of the project 'Ex post estimates of costs to business of EU environmental policies'. Institute for Environmental Studies (IVM), Vrije Universiteit, Amsterdam, Netherlands.

Kumpula, J, Colpaert, A, Tanskanen, A, Anttonen, M, Törmänen, H and Siitari, J (2006) *Porolaidunten inventoinnin kehittäminen*. Keski-Lapin paliskuntien laiduninventointi vuosina 2005-2006. KALA - JA RIISTARAPORTTEJA nro 397, Riistan-ja kalantutkimus, Kaamanen, Helsinki.

Lake, S, Bullock, J M and Hartley, S (2001) *Impacts of livestock grazing on lowland heathland in the UK*. 422, English Nature Research Report, English Nature, Peterborough.

Lal, R (2004) Soil carbon sequestration impacts on global climate and food security. *Science*, No 304, (5677) pp1623-1627.

Landesregierung Sachsen-Anhalt (2009) *Merkblatt Agrarumweltmassnahme Freiwillige Naturschutzleistungen (FNL) Sachsen-Anhalt*. Neuantrag über die Gewährung von Zuwendungen für den Verpflichtungszeitraum 01.10.2009 bis 30.09.2014. Landesregierung Sachsen-Anhalt, Germany.

Lawson, C R, Bennie, J J, Thomas, C D, Hodgson, J A and Wilson, R J (2012) Local and landscape management of an expanding range margin under climate change. *Journal of Applied Ecology*, No 49, (3) pp552-561.

Lawton, J H, Brotherton, P N M, Brown, V K, Elphick, C, Fitter, A H, Forshaw, J, Haddow, R W, Hilborne, S, Leafe, R N, Mace, G M, Southgate, M P, Sutherland, W J, Tew, T E, Varley, J and Wynne, G R (2010) *Making Space for Nature: a review of England's wildlife sites and ecological network.* Report to Defra.

Le Féon, V, Schermann-Legionnet, A, Delettre, Y, Aviron, S, Billeter, R, Bugter, R, Hendrickx, F and Burel, F (2010) Intensification of agriculture, landscape composition and wild bee communities: A large scale study in four European countries. *Agriculture, Ecosystems & Environment*, No 137, (1-2) pp143-150.

Lee, M (2001) Coastal defence and the Habitats Directive: predictions of habitat change in England and Wales. *Geographical Journal*, No 167, (1) pp39-56.

Lesschen, J P, Cammeraat, L H and Nieman, T (2008a) Erosion and terrace failure due to agricultural land abandonment in a semi-arid environment. *Earth Surface Processes and Landforms*, No 33, (10) pp1574-1584.

Lesschen, J P, Schils, R, Kuikman, P and Oudendag, D (2008b) *PICCMAT Climate change mitigation through agricultural techniques. European quantification results.* PICCMAT Deliverable D7. PICCMAT Policy Incentives for Climate Change Mitigation Agricultural Techniques.

Löf, M, Bergquist, J, Brunet, J, Karlsson, M and Torkel Welander, N (2010) Conversion of Norway spruce stands to broadleaved woodland - regeneration systems, fencing and performance of planted seedlings. *Ecological Bulletins*, No 53, pp165-173.

MacArthur, R H and Wilson, E O (1967) *The Theory of Island Biogeography.* Princetown University Press, Princetown, New Jersey, USA.

Macher, C, Guyader, O, Talidec, C and Bertignac, M (2008) A cost benefit analysis of improving trawl selectivity in the case of discards: The *Nephrops norvegicus* fishery in the Bay of Biscay. *Fisheries Research*, No 92, (1) pp76-89.

Macmillan, D C, Harley, D and Morrison, R (1998) Cost-effectiveness analysis of woodland ecosystem restoration. *Ecological Economics*, No 27, pp313-324.

Maes, J, Paracchini, M L and Zulian, G (2011) *A European assessment of the provision of Ecosystem Services - towards an atlas of Ecosystem Services*. JRC Scientific and Technical Reports, Publications Office of the European Union, Luxembourg.

Maes, J, Teller, A, Liquete, C, Erhard, M, Berry, P, Egoh, B, Braat, L, Puydarrieux, P, Fiorina, C, Santos, F, Paracchini, M L, Keune, H, Wittmer, H, Fiala, I, Verburg, P H and Condé, S (2012) Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020, in *Discussion paper draft version 8.3 (25/08/2012),* 3rd European Congress of Conservation Biology. EU Working Group MAES (Mapping and Assessing European Ecosystems and their Services),

Maltby, E (2010) Effects of climate change on the societal benefits of UK upland peat ecosystems: applying the ecosystem approach. *Climate Research*, No 45, pp249-259.

Maltby, E, Ormerod, S, Acreman, M, Blackwell, M, Durance, I, Everard, M, Morris, J, Spray, C J and et al (2011) Chapter 9 Freshwaters - Openwaters, Wetlands and Floodplains, in UK NEA (ed) *The UK National Ecosystem Assessment Technical Report,* UNEP-WCMC, Cambridge.

Markland, J (2002) *Final Report on Financing Natura 2000*. Working Group on Article 8 of the Habitats Directive. European Commission, Brussels.

Martinez Sanchez, S, Ramil Rego, P, Hinojo Sanchez, B and Salinero, E C (2011) Assessing Loss of Biodiversity in Europe Through Remote Sensing:The Necessity of New Methodologies, in O Grillo & G Venora (eds) *Biodiversity Loss in a Changing Planet,* pp19-48. vol. Chapter 2 InTech Open Access,

Martínez, J, Vega-Garcia, C and Chuvieco, E (2009) Human-caused wildfire risk rating for prevention planning in Spain. *Journal of Environmental Management*, No 90, (2) pp1241-1252.

Matthews, A (2013) The CAP budget in the MFF agreement. 3 July 2013. www.CAPreform.eu

Mazza, L, Bennett, G, De Nocker, L, Gantioler, S, Loscarcos, L, Margerison, C, Kaphengst, T, McConville, A J, Rayment, M, ten Brink, P, Tucker, G M and van Diggelen, R (2012) *Green Infrastructure Implementation and Efficiency*. Final report for the European Commission, DG Environment on Contract ENV.B.2./SER/2010/0059., Institute for European Environmental Policy, Brussels and London.

McCorry, M and Ryle, T (2009) *Saltmarsh monitoring project 2007-2008: final report*. A report for Research Branch, National Parks and Wildlife Service, National Parks and Wildlife Service, Ireland.

McCrea-Strub, A, Zeller, D, Rashid Sumaila, U, Nelson, J, Balmford, A and Pauly, D (2011) Understanding the cost of establishing marine protected areas. *Marine Policy*, No 35, (1) pp1-9.

MCPFE, UNECE and FAO (2007) *State of Europe's Forests 2007*. The MCPFE Report on Sustainable Forest Management in Europe. Ministerial Conference on the Protection of Forests in Europe (MCPFE) Liaison Unit, Warsaw.

Medarova-Bergstrom, K and Volkery, A (2012) *Walking the talk - practical options for making the 2014-2020 EU MFF deliver on climate change*. Final Report for the Dutch Ministry of Infrastructure and the Environment, Institute for European Environmental Policy, Brussels.

Metla (2012) *State of Finland's Forests 2010*. Based on the criteria and indicators of sustainable forest management. 5a/2011, Ministry of Agriculture and Forestry & Finnish Forest Research Institute (Metla).

Meulen, F, Bakker, T and Houston, J (2008) The Costs of our Coasts: Examples of Dynamic Dune Management from Western Europe. Martinez, M and Psuty, N. 1[Volume 171], 259-277. Berlin Heidelberg, Springer. Ecological Studies. Caldwell, M. M., Heldmaier, G., Jackson, R. B., Lange, O. L., Mooney, H. A., Schulze, E. D., Sommer, U., Caldwell, M. M., Heldmaier, G., Jackson, R. B., Lange, O. L., Mooney, H. A., Schulze, E. D., and Sommer, U.

Millennium Ecosystem Assessment (2005) *Ecosystems and Human Well-being: Synthesis*. World Resources Institute, Washington, DC.

Minayeva, T, Sirin, A and Bragg, O (2009) *A Quick Scan of Peatlands in Central and Eastern Europe*. 132 pp, tabl. 6, fig. 17., Wetlands International, Wageningen, The Netherlands.

Ministry of Rural Development and Food (2012) Rural Development Programme for Greece. Hellenic Republic.

MMM (2011) *Ehdotus soiden ja turvemaiden kestävän ja vastuullisen käytön ja suojelun kansalliseksi strategiaksi (Finland Peatland Strategy)*. MMM 2011:1, MMM Työryhmämuistio, Helsinki.

Montanarella, L, Jones, R J A and Hiederer, R (2006) The distribution of peatland in Europe. *Mires and Peat*, No 1, Article 1.

Montoya, D, Rogers, L and Memmott, J (2012) Emerging perspectives in the restoration of biodiversity-based ecosystem services. *TRENDS in Ecology and Evolution*, No 27, (12) pp666-672.

Moreira, F, Viedma, O, Arianoutsou, M, Curt, T, Koutsias, N, Rigolot, E, Barbati, A, Corona, P, Vaz, P, Xanthopoulos, G, Mouillot, F and Bilgili, E (2011) Landscape-wildfire interactions in

Southern Europe: implications for landscape management. *Journal of Environmental Management*, No 92, (10) pp2389-2402.

Moreno-Mateos, D, Power, M E, Comín, F A and Yockteng, R (2012) Structural and functional loss in restored wetland ecosystems. *PLoS Biol*, No 10, (1) e1001247.

Morling, P (2008) *Funding for Biodiversity: An analysis of the UK's spend towards the 2010 target*. IUCN UK Committee, Peterborough.

Morris, R K A, Alonso, I, Jefferson, R G and Kirby, K J (2006) The creation of compensatory habitat - Can it secure sustainable development? *Journal for Nature Conservation*, No 14, (2) pp106-116.

Moss, B, Hering, D, Green, A J, Aidoud, A, Becares, E, Beklioglu, M, Bennion, H, Boix, D, Brucet, S, Carvalho, L, Clement, B, Davidson, T, Declerck, S, Dobson, M, van Donk, E, Dudley, B, Feuchtmayr, H, Friberg, N, Grenouillet, G, Hillebrand, H, Hobaek, A, Irvine, K, Jeppesen, E, Johnson, R, Jones, I, Kernan, M, Lauridsen, T L, Manca, M, Meerhoff, M, Olafsson, J, Ormerod, S, Papastergiadou, E, Penning, W E, Ptacnik, R, Quintana, X, Sandin, L, Seferlis, M, Simpson, G, Triga, C, Verdonschot, P, Verschoor, A M and Weyhenmeyer, G A (2009) Climate change and the future of freshwater biodiversity in Europe: A primer for policy-makers. *Freshwater Reviews*, No 2, (2) pp103-130.

Mossman, H L, Davy, A J and Grant, A (2012) Does managed coastal realignment create saltmarshes with 'equivalent biological characteristics' to natural reference sites? *Journal of Applied Ecology*, No 49, (6) pp1446-1456.

Muller, J (2004) Cost-benefit ratio and empirical examination of the acceptance of heathland maintenance in the Lueneburg heath nature reserve. *Journal of Environmental Planning and Management*, No 47, (5) pp757-771.

Munoz-Perez, J J, Lopez de San Roman-Blanco, B, Gutierrez-Mas, J M, Moreno, L and Cuena, G J (2001) Cost of beach maintenance in the Gulf of Cadiz (SW Spain). *Coastal Engineering*, No 42, (2) pp143-153.

Murphy, P (2008) *Restoring Active Blanket Bog in Ireland*. LIFE Technical Final Report. LIFE02 NAT/IRL/8490.

Naidoo, R, Balmford, A, Ferraro, P J, Polasky, S, Ricketts, T H and Rouget, M (2006) Integrating economic costs into conservation planning. *Trends in Ecology & Evolution*, No 21, (12) pp681-687.

National Parks and Wildlife Service (2008) *The status of EU protected habitats and species in Ireland*. National Parks and Wildlife Service, Dublin, Ireland.

Natural England (2008) *State of the Natural Environment 2008*. NE85, Natural England, Sheffield.

Natural England (2009) *Higher Level Stewardship: Environmental Stewardship Handbook*. Natural England.

Natural England (2010) *England's peatlands: carbon storage and greenhouse gasses*. NE257, Natural England, Peterborough.

Natural England (2011) *Aggregates Levy Sustainability Fund Report 2008-2011*. Natural England.

Naumann, S, McKenna, D, Kaphengst, T, Pieterse, M and Rayment, M (2011) *Design, Implementation and Cost Elements of Green Infrastructure Projects*. Final report for the European Commission, DG Environment, Contract no. 070307/2010/577182/ETU/F, Ecologic Ltd and GHK Consulting.

Nellemann, C and Corcoran, E (2010) *Dead Planet, Living Planet - Biodiversity and ecosystem restoration for sustainable development*. A rapid response assessment. UNEP, GRID-Arendal, www.grida.no.

Newton, A C, Stewart, G B, Myers, G, Diaz, A, Lake, S, Bullock, J M and Pullin, A S (2009) Impacts of grazing on lowland heathland in north-west Europe. *Biological Conservation*, No 142, (5) pp935-947.

Nieto, A and Alexander, K N A (2010) *European Red List of Saproxylic Beatles*. Publications Office of the European Union, Luxembourg.

Normander, B, Levin, G, Auvinen, A-P, Bratli, H, Stabbetorp, O, Hedblom, M, Glimskär, A and Gudmundsson, G A (2009) *State of biodiversity in the Nordic countries. An assessment of progress towards achieving the target of halting biodiversity loss by 2010.* 509, Nordic Council of Ministers.

Nottage, A S and Robertson, P A (2005) *The saltmarsh creation handbook: a project manager's guide to the creation of saltmarsh and intertidal mudflat*. The RSPB and CIWEM, Sandy & London.

Nowicki, P, Goba, V, Knierim, A, van Meijl, H, Banse, M, Delbaere, B, Helming, J, Hunke, P, Jansson, K, Jansson, T, Jones-Walters, L, Mikos, V, Sattler, C, Schlaefke, N, Terluin, I and Verhoog, D (2009) *Scenar 2020-II. Update of Analysis of Prospects in the Scenar 2020 Study - Preparing for Change*. Final Report. Contract No. 30-CE-0200286/00-21, European Commission DG Agriculture and Rural Development, Brussels.

Olmeda, C, Keenleyside, C, Tucker, G M and Underwood, E (2013) *Farming for Natura 2000. Guidance on how to integrate Natura 2000 conservation objectives into farming practices based on Member States good practice experiences*. European Commission, Brussels.

Oltmer, K, van Bommel, K H M, Clement, J, Rudrum, D P and Schouwneberg, E P A G (2009) *Kosten voor habitattypen in Natura 2000-gebieden. Toepassing van de methode Kosteneffectiviteit natuurbeleid*. Werkdocument 152, Wageningen.

Oppermann, R, Beaufoy, G and Jones, G (eds) (2012) *High Nature Value Farming in Europe. 35 European Countries - Experiences and Perspectives.* verlag regionalkultur, Ubstadt-Weiher.

Özden, Ö 2009. *The Biodiversity of Invertebrates in Cyprus Ecosystems.* Doctor of Philosophy in Biological Sciences University of Exeter.

Paavilainen, E and Päivänen, J (1995) Peatland Forestry - Ecology and Principles. [111]. Berlin, Springer. Ecological Studies.

Pain, D J and Pienkowski, M W (eds) (1997) *Farming and Birds in Europe. The Common Agricultural Policy and Its Implications for Bird Conservation.* Academic Press, London.

Pakalne, M (2004) *Wetland management methods in protected nature areas and their application in LIFE-Nature projects*. Background paper for the LIFE-Nature Co-op project "Experience exchange on habitat management among Baltic LIFE-Nature projects". Baltic Environmental Forum BEF.

Papanastasis, V P, Kyriakakis, S and Kazakis, G (2002) Plant diversity in relation to overgrazing and burning in mountain mediterranean ecosystems. *Journal of Mediterranean Ecology*, No 3, (2-3) pp53-63.

Papanikolaou, A D, Fyllas, N M, Mazaris, A D, Dimitrakopoulos, P G, Kallimanis, A S and Pantis, J D (2011) Grazing effects on plant functional group diversity in Mediterranean shrublands. *Biodiversity and Conservation*, No 20, (12) pp2831-2843.

Peltomaa, R (2007) Drainage of forests in Finland. *Irrigation and Drainage*, No 56, (Supplement 1) S151-S159.

Pérez-Ruzafa, A, Marcos, C and Pérez-Ruzafa, I M (2011) Mediterranean coastal lagoons in an ecosystem and aquatic resources management context. *Physics and Chemistry of the Earth*, No 36, (5-6) pp160-166.

Perrow, M R and Davy, A J (2002) *Handbook of Ecological Restoration.* Cambridge University Press, Cambridge.

Peterken, G (1993) *Woodland Conservation and Management.* 2nd ed. Chapman & Hall, London.

Petley, S (2012) *Exploring the use of forest bonds to support woodland creation*. A contribution to the Finance Workstream of the Woodland Carbon Task Force, Forestry Commission England.

Picchi, S (2008) *Management of Natura 2000 Habitats: 2250 Coastal dunes with Juniperus spp.* Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 06/24, European Commission, Brussels.

Pickaver, A and Ferreira, M (2008) *Implementing ICZM at sub-national /local level - recommendations on best practice*. Interreg COREPOINT final report. EUCC - The Coastal Union.

Plieninger, T, Höchtl, F and Spek, T (2006) Traditional land-use and nature conservation in European rural landscapes. *Environmental Science & Policy*, No 9, (4) pp317-321.

Poláková, J, Tucker, G M, Hart, K, Dwyer, J and Rayment, M (2011) *Addressing biodiversity and habitat preservation through Measures applied under the Common Agricultural Policy*. Report prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0388497/00-44, Institute for European Environmental Policy, London.

Provoost, S, Jones, M L M and Edmondson, S E (2011) Changes in landscape and vegetation of coastal dunes in northwest Europe: a review. *Journal of Coastal Conservation*, No 15, (1) pp207-226.

Pryor, S N and Jackson, T J F (2002) *The cost of restoring plantations on ancient woodland sites*. The Woodland Trust, England.

Pywell, R F, Bullock, J M, Hopkins, A, Walker, K J, Sparks, T H, Burke, M J W and Peel, S (2002) Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. *Journal of Applied Ecology*, No 39, (2) pp294-309.

Ramchunder, S, Brown, L E and Holden, J (2012) Catchment-scale peatland restoration benefits stream ecosystem biodiversity. *Journal of Applied Ecology*, No 49, (1) pp182-191.

Ramón Vallejo, V, Arianoutsou, M and Moreira, F (2012) Fire Ecology and Post-Fire Restoration Approaches in Southern European Forest Types, in F Moreira, M Arianoutsou, P Corona, & J De las Heras (eds) *Post-Fire Management and Restoration of Southern European Forests,* Springer,

Ramos, M C, Cots-Folch, R and Martínez-Casasnovas, J A (2007) Sustainability of modern land terracing for vineyard plantation in a Mediterranean mountain environment – The case of the Priorat region (NE Spain). *Geomorphology*, No 86, (1-2) pp1-11.

Raunio, A, Schulman, A and Kontula, T (2008) *Assessment of Threatened Habitat Types in Finland*. [Suomen luontotyyppien uhanalaisuus - Osa 1 Tulokset ja arvioinnin perusteet]. Suomen ympäristö 8/2008, Suomen ympäristökeskus [The Finnish Environment Institute], Helsinki.

Rey Benayas, J M, Newton, A C, Diaz, A and Bullock, J M (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, No 325, (5944) pp1121-1124.

Rienks, W A (2008) *The future of rural Europe. An anthology based on the results of the Eururalis 2.0 scenario study.* 1, W A Rienks (ed), Wageningen University Reasearch and Netherlands Environmental Assessment Agency., Wageningen, The Netherlands..

Rigueiro-Rodríguez, A, McAdam, J and Mosquera-Losada, M R (2009) *Agroforestry in Europe.* Springer,

RIKZ and Roelse, P (2002) *Water en zand in Balans (water and sand in balance)*. Evaluatie zandsuppleties na 1990 een morfologische beschouwing (*evaluation of sand supply after 1990 a morphological consideration*). 2002.003, RIKZ.

Robledano, F, Esteve, M A, Farinós, P, Carreño, M F and Martínez-Fernández, J (2010) Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands. *Ecological Indicators*, No 10, (2) pp274-286.

Rodriguez, J P, Beard, T D, Bennett, E M, Cumming, G S, Cork, S J, Agard, J, Dobson, A P and Peterson, G D (2006) Trade-offs across space, time, and ecosystem services. *Ecology and Society*, No 11, (28) [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art28/.

RSPB (2010) *Supporting Business for Biodiversity: A biodiversity biodiversity technical assistance facility. Final Report*. European Commission: EC contract 070501/2006/451431/MA/B2.

RSPB (2011) *LIFE Active Blanket Bog in Wales*. LIFE06 NAT/UK/000134, R Farmer (ed), RSPB.

RSPB, Birdlife International, T&E, EEB and CEE Bankwatch Network (2008) *TEN-T and Natura 2000: The way forward - An assessment of the potential impact of the TEN-T priority projects on Natura 2000*. Birdlife International.

Ruiz, E (2008) *Management of Natura 2000 Habitats: 3170 Mediterranean temporary ponds*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 07/24, European Commission, Brussels.

Rupp, S and Nicholls, R J (2002) *Managed realignment of coastal flood defences: a comparison between England and Germany*. Unpublished report prepared for proceedings of "Dealing with Flood Risk" An interdisciplinary seminar of the regional implications of modern flood management. Bob van Kappel (ed.), Delft Hydraulics. 4 March 2002.

Russi, D, ten Brink, P, Farmer, A, Badura, T, Coates, D, Förster, J, Kumar, R and Davison, N (2013) *The Economics of Ecosystems and Biodiversity for Water and Wetlands*. Ramsar Secretariat, http://www.teebweb.org/wetlands/.

RWS Waterdiens (2012) Helpdesk Water Netherlands. RWS Waterdiens http://www.helpdeskwater.nl/onderwerpen/water-ruimte/ecologie_maatregelen/ingreep_in_de_chemie/baggeren/

Salman, A, Lombardo, S and Doody, P (2004) *Living with coastal erosion in Europe: Sediment and space for sustainability (PART 1)*. PART I - Major findings and Policy Recommendations of the EUROSION project. Service contract B4-3301/2001/329175/MAR/B3 "Coastal erosion - Evaluation of the need for action" for DG Environment, European Commission.

San Miguel, A (2008) *Management of Natura 2000 Habitats: 6220 Pseudo-steppe with grasses and annuals (Thero-Brachypodietea)*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 13/24, European Commission, Brussels.

Santilli, E, Lombardo, L, Varlaro, M E, Nannelli, R, Gagnarli, E and Bati, C B (2011) Effectiveness of the GAEC cross-compliance standard Maintenance of olive groves in good vegetative condition in avoiding the deterioration of habitats and land abandonment. *Italian Journal of Agronomy*, No 6, (s1) e15.

Scalera, R (2008) *EU funding for management and research of invasive alien species in Europe*. Final report for the EEA (Contract number 3603/B2007.EEA.53070), European Environment Agency, Copenhagen.

Scalera, R (2010) How much is Europe spending on invasive alien species? *Biological Invasions*, No 12, (1) pp173-177.

Schäfer, A (2009) Moore und Euros - die vergessenen Millionen. *Archiv fuer Forstwesen und Landschaftsoekologie*, No 43, (4) pp156-161.

Schaub, M, Martinez, N, Tagmann-Ioset, A, Weisshaupt, N, Maurer, M L, Reichlin, T S, Abadi, F, Zbinden, N, Jenni, L and Arlettaz, R (2010) Patches of bare ground as a staple commodity for declining ground-foraging insectivorous farmland birds. *PLoS ONE*, No 5, (10) e13115.

Schramek, J, Osterburg, B, Kasperczyk, N, Nitsch, H, Wolff, A, Weis, M and Hülemeyer, K (2012) *Vorschläge zur Ausgestaltung von Instrumenten für einen effektiven Schutz von Dauergrünland*. [recommendations for the development of instruments to protect permanent grassland]. Bundesamt für Naturschutz, Bonn, Germany.

Schuna, B (1999) Techniques for heathland restoration in England. *Restoration and Reclamation review*, No 4, (2)

Scottish Natural Heritage (2000) *A guide to managing coastal erosion in beach/dune ecosystems*. Scottish Natural Heritage.

Scottish Natural Heritage (2011) *The Fen Management Handbook*. A McBride, I Diack, N Droy, B Hamill, P Jones, J Schutten, A Skinner, & M Street (eds), Scottish Natural Heritage, Edinburgh.

Šeffer, J, Janák, M and Šefferová, S V (2008) *Management of Natura 2000 Habitats: 6440 Alluvial meadows of river valleys of the Cnidion dubii*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 17/24, European Commission, Brussels.

Šefferová, S V, Šeffer, J and Janák, M (2008) *Management of Natura 2000 Habitats: 7230 Alkaline fens*. Technical Report 2008 20/24, European Commission, Brussels.

SERI (2004) *SER International Primer on Ecological Restoration*. SERI Science & Policy Working Group. Society for Ecological Restoration International, Tucson.

Shaw, S C, Wheeler, B D, Kirby, P, Phillipson, P and Edmunds, R (1996) *Literature review of the historical effects of burning and grazing of blanket bog and upland wet heath*. 172, English Nature Research Report, English Nature, Peterborough.

Shine, C, Kettunen, M, ten Brink, P, Genovesi, P and Gollasch, S (2009) *Technical support to EU strategy on invasive alien species (IAS) - Recommendations on policy options to control the negative impacts of IAS on biodiversity in Europe and the EU*. (final report for the European Commission). Institute for European Environmental Policy, Brussels, Belgium.

Shine, C, Kettunen, M, Genovesi, P, Essi, F, Gollasch, S, Rabitsch, W, Scalera, R, Starfinger, U and ten Brink, P (2010) *Assessment to support continued development of the EU Strategy to combat invasive alien species*. Final Report for the European Commission, Institute for European Environmental Policy (IEEP), Brussels.

Similä, M, Junninen, K and (eds.) (2012) *Ecological restoration and management in boreal forests*. Best practices from Finland. Metsähallitus, Finland.

Slingenberg, A, Braat, L, van der Windt, H, Raedemakers, K, Eichler, L and Turner, K (2009) *Study on understanding the causes of biodiversity loss and the policy assessment framework*. Report for European Commission DG Environment in the context of the Framework Contract No. DG ENV/G.1/FRA/2006/0073, ECORYS Member of Consortium COWI, Rotterdam.

Smith, P and Olesen, J E (2010) Synergies between the mitigation of, and adaptation to, climate change in agriculture. *Journal of Agricultural Science*, No 148, (5) pp543-552.

Smith, P, Martino, D, Cai, Z, Gwary, D, Janzen, H, Kumar, P, McCarl, B, Ogle, S, O'Mara, F, Rice, C, Scholes, B, Sirotenko, O, Howden, M, McAllister, T, Pan, G, Romanenkov, V, Schneider, U, Towprayoon, S, Wattenbach, M and Smith, J (2008) Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society of London.Series B: Biological Sciences*, No 363, (1492) pp789-813.

Spencer, K L and Harvey, G L (2012) Understanding system disturbance and ecosystem services in restored saltmarshes: Integrating physical and biogeochemical processes. *Estuarine, Coastal and Shelf Science*, No 106, pp23-32.

Stallegger, M (2008) *Management of Natura 2000 Habitats: 7150 Depression on peat substrates of the Rhynchosporion*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 19/24, European Commission, Brussels.

Stoate, C, Báldi, A, Beja, P, Boatman, N D, Herzon, I, van Doorn, A, de Snoo, G R, Rakosy, L and Ramwell, C (2009) Ecological impacts of early 21st century agricultural change in Europe – A review. *Environmental Management*, No 91, (1) pp22-46.

Strange, N, Jacobsen, J, Thorsen, B and Tarp, P (2007) Value for Money: Protecting Endangered Species on Danish Heathland. *Environmental Management*, No 40, (5) pp761-774.

Stunell, J (2010) *Investigating the impacts of windfarm development on peatlands in England*. NECR032, Natural England, Peterborough.

Suárez-Seoane, S, Osborne, P E and Baudry, J (2002) Responses of birds of different biogeographic origins and habitat requirements to agricultural land abandonment in northern Spain. *Biological Conservation*, No 105, (3) pp333-344.

Sutcliffe, O L and Kay, Q O N (2000) Changes in the arable flora of central southern England since the 1960s. *Biological Conservation*, No 93, (1) pp1-8.

Symes, A 2006. *Is the abandonment of traditional low-intensity agriculture detrimental to farmland birds in Cyprus?* MSc University of East Anglia.

TEEB (2008) *The Economics of Ecosystems and Biodiversity: An Interim Report*. European Commission, Brussels.

TEEB (2010a) *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.* Earthscan, London and Washington, DC.

TEEB (2010b) *The Economics of Ecosystems and Biodiversity for Local and Regional Policy Makers.* Earthscan, London and Washington, DC.

TEEB (ed) (2011) *The Economics of Ecosystems and Biodiversity in National and International Policy Making. Edited by Patrick Ten Brink.* Earthscan, London and Washington.

ten Brink, P, Badura, T, Bassi, S, Daly, E, Dickie, I A, Ding, H, Gantioler, S, Gerdes, H, Kettunen, M, Lago, M, Lang, S, Markandya, A, Nunes, P A L D, Pieterse, M, Rayment, M and Tinch, R (2011) *Estimating the overall economic value of the benefits provided by the Natura 2000 network*. Final report to the European Commission, DG Environment on Contract ENV.B.2/SER/2008/0038, Institute for European Environmental Policy / GHK / Ecologic, Brussels.

Tena, A, Batalla, R J, Vericat, D and López-Tarazón, J A (2011) Suspended sediment dynamics in a large regulated river over a 10-year period (the lower Ebro, NE Iberian Peninsula). *Geomorphology*, No 125, (1) pp73-84.

Thauront, M and Stallegger, M (2008) *Management of Natura 2000 Habitats: 9110 Luzulo-Fagetum beech forests*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 22/24, The European Commission.

The River Restoration Centre (2002) *Manual of River Restoration Techniques*. The River Restoration Centre, Cranfield.

The Scottish Government (2012) Standard Cost Capital Items for Rural Development Contracts - Rural Priorities. http://www.scotland.gov.uk/Topics/farmingrural/SRDP/RuralPriorities/RPACPaymentRate/StandardCostCapital

Thomas, F, Denzel, K, Hartmann, E, Luick, R and Schmoock, K (2009) *Kurzfassungen der Agrarumwelt- und Naturschutzprogramme in der Bundesrepublik Deutschland*. Darstellung und Analyse der Entwicklung von Massnahmen der Agrarumwelt- und Naturschutzprogramme in der Bundesrepublik Deutschland. BfN - Skripten 253, Bundesamt für Naturschutz, Bonn-Bad Godesberg, Germany.

Thompson, R, Humphrey, J, Harmer, R and Ferris, R (2003) *Restoration of Native Woodland on Ancient Woodland Sites: Practice Guide*. The Forestry Commission, Edinburgh.

Tockner, K, Robinson, C T and Uehlinger, U (eds) (2009) *Rivers of Europe.* First edition ed. Academic Press, London.

Török, K, Szili-Kovács, T, Halassy, M, Tóth, T, Hayek, Zs, Paschke, M W and Wardell, L J (2000) Immobilization of soil nitrogen as a possible method for the restoration of sandy grassland. *Applied Vegetation Science*, No 3, (1) pp7-9.

Török, P, Vida, E, Deák, B, Lengyel, S and Tóthmérész, B (2011) Grassland restoration on former croplands in Europe: an assessment of applicability of techniques and costs. *Biodiversity and Conservation*, No 20, (11) pp2311-2332.

Triisberg, T, Karofeld, E and Paal, J (2011) Re-vegetation of block-cut and milled peatlands: an Estonian example. *Mires and Peat*, No 8, Article 5.

Tucker, G M (2003) *Review of the Impacts of Heather and Grassland Burning in the Uplands on Soils, Hydrology and Biodiversity*. 550, English Nature, Peterborough.

Tucker, G M and Evans, M (1997) *Habitats for Birds in Europe: a Conservation Strategy for the Wider Environment.* BirdLife International, Cambridge.

Turbé, A, De Toni, A, Benito, P, Lavelle, P, Lavelle, P, Ruiz, N, Van der Putten, W H, Labouze, E and Mudgal, S (2010) *Soil biodiversity: functions, threats and tools for policy makers*. [Contract 07.0307/2008/517444/ETU/B1], Report for European Commission (DG Environment), Bio Intelligence Service, IRD, and NIOO, Luxembourg.

Turner, R K, Burgess, D, Hadley, D, Coombes, E and Jackson, N (2007) A cost-benefit appraisal of coastal managed realignment policy. *Global Environmental Change*, No 17, (3-4) pp397-407.

UK Biodiversity Group (2000) *Tranche 2 action plans cost estimates - maritime habitats*.

UK NEA (2011a) Chapter 6 Semi-Natural Grasslands, in UK NEA (ed) *The UK National Ecosystem Assessment Technical Report,* UNEP-WCMC, Cambridge.

UK NEA (2011b) Chapter 11 Coastal margins, in *The UK National Ecosystem Assessment Technical Report,* UNEP-WCMC, Cambridge.

van Andel, J and Aronson, J (2006) *Restoration Ecology: the New Frontier.* Wiley-Blackwell,

van den Broek, J A, van Hofwegen, G, Beekman, W and Woittez, M (2007) *Options for increasing nutrient use efficiency in Dutch dairy and arable farming towards 2030: an exploration of cost-effective measures at farm and regional levels*. WOt-rapport No.55, Statutory Research Tasks Unit for Nature & the Environment, Wageningen.

van der Bolt, F J E, van Boekel, E M P M, Clevering, O A, van Dijk, W, Hoving, I E, Kselik, R A L, de Klein, J J M, Leenders, T P, Linderhof, V G M, Massop, H T L, Mulder, H M, Noij, G J, van Os, E A, Polman, N B P, Renaud, L V, Schouwmans, O F and Walvoort, D J J (2008) *Ex-ante evaluatie landbouw en KRW; Effecten van voorgenomen en potentieel aanvullende maatregelen op de oppervlaktewaterkwaliteit voor nutriënten*. Alterra-rapport 1687, Alterra, Wageningen.

van Swaay, C, Warren, M and Lois, G (2006) Biotope use and trends of European butterflies. *Journal of Insect Conservation*, No 10, (2) pp189-209.

Vandewalle, M, Sykes, M T, Harrison, P A, Luck, G W, Berry, P, Bugter, R, Dawson, T P, Feld, C K, Harrington, R, Haslett, J R, Hering, D, Jones, K B, Jongman, R, Lavorel, S, Martins da Silva, P, Moora, M, Paterson, J, Rounsevell, M D A, Sandin, L, Settele, J, Sousa, J P and Zobel, M (2010) *Review paper on concepts of dynamic ecosystems and their services*. The RUBICODE project - Rationalising Biodiversity Conservation in Dynamic Ecosystems. Funded under the European Commission Sixth Framework Programme (Contract Number: 036890), RUBICODE project.

Vasander, H, Tuittila, E-S, Lode, E, Lundin, L, Ilomets, M, Sallantaus, T, Heikkilä, R, Pitkänen, M-L and Laine, J (2003) Status and restoration of peatlands in northern Europe. *Wetlands Ecology and Management*, No 11, (1-2) pp51-63.

Verheijen, F G A, Jones, R J A, Rickson, R J and Smith, C J (2009a) Tolerable versus actual soil erosion rates in Europe. *Earth-Science Reviews*, No 94, (1-4) pp23-38.

Verheijen, L, Post, R, Mulder, T, van der Reest, P J, Hummelen, L A, Bos, J, de Boer, T F, van der Zee, F and Bal, D (2009b) *Advies commissie-Verheijen herberekening standaardkostprijzen natuurbeheer*. IPO/ LNV.

Verhulst, J, Báldi, A and Kleijn, D (2004) Relationship between land-use intensity and species richness and abundance of birds in Hungary. *Agriculture, Ecosystems & Environment*, No 104, (3) pp465-473.

Verkerk, P J, Zanchi, G and Lindner, M (2008) *Impacts of biological and landscape diversity protection on the wood supply in Europe*. EFI Technical Report 27, European Forest Institute, Joensuu Finland.

Verkerk, P J, Lindner, M, Zanchi, G and Zudin, S (2011) Assessing impacts of intensified biomass removal on deadwood in European forests. *Ecological Indicators*, No 11, (1) pp27-35.

VV.AA (2009) Bases Ecológicas Preliminares Para La Conservación De Los Tipos De Hábitat De Interés Comunitario En España. Ministerio de Medio Ambiente, y Medio Rural y Marino online resource http://www.magrama.gob.es/es/biodiversidad/temas/red-natura-2000/documentos-claves-de-la-red-natura-2000/acceso_fichas2010-10-28_21.04.13.6872.aspx

Wätzold, F and Schwerdtner, K (2005) Why be wasteful when preserving a valuable resource? A review article on cost-effectiveness of European biodiversity conservation policy. *Biological Conservation*, No 123, (3) pp327-338.

Westhoek, H, van Zeijts, H, Witmer, M, van den Berg, M, Overmars, K, van der Esch, S and van der Bilt, W (2012) *Greening the CAP: An analysis of the effects of the European Commission's proposals for the Common Agricultural Policy 2014–2020*. PBL Netherlands Environmental Assessment Agency, The Hague.

Wilby, R L, Orr, H G, Hedger, M, Forrow, D and Blackmore, M (2006) Risks posed by climate change to the delivery of Water Framework Directive objectives in the UK. *Environment International*, No 32, (8) pp1043-1055.

Wildlife Trusts Wales (2011) *Pumlumon Project Expected Outcomes and Deliveries*.

Wilson, D, Renou-Wilson, F, Farrell, C, Bullock, C and Müller, C (2012) *Carbon Restore - The potential of restored Irish peatlands for carbon uptake and storage*. The potential of peatlands for carbon sequestration. CCRP Report Series No.15, Environmental Protection Agency, Ireland.

Wilson, E and Piper, J (2008) Spatial planning for biodiversity in Europe's changing climate. *European Environment*, No 18, (3) pp135-151.

Wolters, M, Garbutt, A and Bakker, J P (2005) Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. *Biological Conservation*, No 123, (2) pp249-268.

Worrall, S (2005) The UK LIFE project on shoreline management: 'Living with the Sea', in J L Herrier & et al (eds) *Proceedings 'Dunes and Estuaries 2005',* pp451-459. International Conference on Nature Restoration Practices in European Coastal Habitats, 19.9.2005, Koksijde, Belgium. VLIZ Special Publication,

Wright, H A and Bailey, A W (1982) *Fire Ecology.* John Wiley & Sons, New York.

WWF (2001) *Insight into Europe's Forest Protection*. World Wide Fund for Nature, Switzerland.

Wye Valley AONB (2009) *A report on Himalayan Balsam control in the Wye valley Area of Outstanding Natural Beauty (AONB)*. Report on trials carried out between 2005 and 2008. Defra, UK.

Zaghi, D (2008a) *Management of Natura 2000 Habitats: 9530 (Sub)-Mediterranean pine forests with endemic black pines*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 24/24, The European Commission, Brussels.

Zaghi, D (2008b) *Management of Natura 2000 Habitats: 4060 Alpine and Boreal heaths*. Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Technical Report 2008 09/24, European Commission, Brussels.

Zdruli, P, Jones, R J A and Montanarella, L (2004) *Organic matter in the soils of Southern Europe*. European Soil Bureau Technical Report EUR 21083 EN, Office for Official Publications of the European Communities, Luxembourg.