JRC Scientific and Technical Reports

Review of published climate change adaptation and mitigation measures related with water

Tiina Nõges, Peeter Nõges, Ana Cristina Cardoso

EUR 24682 EN - 2010

JRC
EUROPEAN COMMISSION

Institute for Environment and Sustainability

The mission of the JRC-IES is to provide scientific-technical support to the European Union's policies for the protection and sustainable development of the European and global environment.

European Commission
Joint Research Centre
Institute for Environment and Sustainability

Contact information
Address: Via E. Fermi 2749, 21027 Ispra (VA), Italy
E-mail: ana-cristina.cardoso@jrc.ec.europa.eu
Tel.: +390332785702
Fax: +390332789352

http://ies.jrc.ec.europa.eu/
http://www.jrc.ec.europa.eu/

JRC 62545

EUR 24682 EN
ISBN 978-92-79-18984-5
ISSN 1018-5593
doi:10.2788/18203

Luxembourg: Publications Office of the European Union

Printed in Italy

Review of published climate change adaptation and mitigation measures related with water

Tiina Nõges, Peeter Nõges, Ana Cristina Cardoso

TABLE OF CONTENTS

List of Abbreviations

AET	Actual evapotranspiration
BMP	Best Management Practice
BOD	Biochemical oxygen demand
CC	climate change
CD	Controlled Drainage
CDM-AR	Clean Development Mechanism-Afforestation/Reforestation
CIS	Common Implementation Strategy
CPOM	Coarse particulate organic matter
DOC	Dissolved organic carbon
ECRR	European Centre for River Restoration
ED	Extended detention (basin)
EEA	European Environment Agency
GCM	Global Climate Model
GDE	Groundwater-dependent ecosystems
GHG	greenhouse gases
GPP	Gross primary production
GWDTE	Groundwater dependent terrestrial ecosystems
ICPDR	International Commission for the Protection of the Danube River
IPCC	Intergovernmental Panel on Climate Change
IRBM	Integrated river basin management
IWMI	International Water Management Institute
IWRM	Integrated Water Resources Management
JRC	Joint Research Centre of European Commission
LID	Low Impact Development
LIUDD	Low Impact Urban Design and Development
MDGs	Millennium Development Goals
MUFS	Multiple Uses and Functions of Water Services
NRRSS	National River Restoration Science Synthesis
PRC	Peak Runoff Control
RBMP	River Basin Management Plan
SAC	Special Area of Conservation
SEPA	Scottish Environment Protection Agency
SGDE	Subsurface groundwater-dependent ecosystems
SS	Suspended Solids
SUDS	Sustainable Urban Drainage Systems
WCE	Western and Central Europe
WDI	Water deficit index
WFD	Water Framework Directive
WSP	Water Safety Plan
WSUD	Water Sensitive Urban Design

Executive summary

European water bodies are already suffering from a number of human activities, such as physical modifications, water abstraction, pollution with nutrients, heat and hazardous substances. Where their conditions allow, they are still been used for fisheries, transport, energy production, and recreational activities. The effects of climate change (CC) are already clearly manifested in some water related aspects like the seasonal flow patterns in rivers, stratification and water level regimes in lakes, frequency of extreme events (floods and low flow), or phenological changes in aquatic foodchains and much bigger changes are expected in the near future. The last report by the International Panel on Climate Change (IPCC) declares it unequivocal that the world is heating up beyond any natural cyclical variations, and that there is 90 per cent certainty that the phenomenon of climate change is caused by humans. Mitigation and adaptation form a two-pronged strategy for dealing with climate change causes and consequences.

Climate mitigation is any action taken to reduce or eliminate the long-term risk and hazards of climate change to human life and property. The IPCC defines mitigation as: "An anthropogenic intervention to reduce the sources or enhance the sinks of greenhouse gases".

Climate adaptation refers to the ability of a system to adjust to climate change (including climate variability and extremes) to moderate potential damage, to take advantage of opportunities, or to cope with the consequences. The IPCC defines adaptation as the "adjustment in natural or human systems to a new or changing environment. Adaptation to climate change refers to adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities. Various types of adaptation can be distinguished, including anticipatory and reactive adaptation, private and public adaptation, and autonomous and planned adaptation."

The purpose of this report is to review the information available in reports and scientific literature about **potential or planned water related measures tackling climate change causes and consequences**. In this context, **measures** are defined as practical steps or actions taken to (i) reduce the sources or enhance the sinks of greenhouse gases, (ii) to decrease the vulnerability of water resources and aquatic ecosystems to climate change, or (iii) enhance the knowledge base on climate-water relationships and increase the capacity of the society to take right decisions on this matter.

The report **does not review** climate change impacts not addressed by specific measures and the issues of natural adaptation in aquatic ecosystems.

The database of measures analysed in this report and given in a separate Annex as an Excel spreadsheet constitutes the most important part of this deliverable. The collection of about 450 measures is a compilation from various published sources, the bulk, however, being collected from the River Basin

Management Plans (RBMPs) during their initial preparation phase (the WRc report - Nixon, 2008) and after adoption (Nõges et al., 2010). In addition, measures have been included from national climate change adaptation plans, published papers and Internet sources.

The list of measures can be filtered by economic sector (8 categories), type of intervention (7), main purpose (8), water category (5), pressure addressed (8), precaution level (4), and country. Additionally measures are labelled by the five specific adaptation strategies addressed in the REFRESH Project (listed further). Text fields (name of the measure, characterise the type of measure as defined in original publication, benefit in current climate, contribution to adaptation to climate change, and potential problems under changing climate) can be used for string search.

In many cases identical or similar measures have been described or proposed by different countries. We did not, however, merge them in order to keep the traceability and the specific information submitted together with the measure description.

25 most relevant measures are described in special text boxes in a form of short encyclopaedia papers following a uniform structure. For each measure the following aspects are described:

1. **Description of the problem** describes the main issues why taking measures has become necessary and shows the linkages of the issue with climate change and other anthropogenic factors.
2. **Description of the measure** shows the principle of intervention and opens the technical details.
3. **Benefit in current climate**. As the majority of measures are designed not directly to cope with climate change, but for other water management issues, their efficiency in present climate conditions is described.
4. **Contribution to adaptation to climate change** describes the expected effect in lowering the vulnerability of natural or human systems and the synergies with other mitigation and adaptation measures.
5. **Potential conflicts and problems under changing climate**, precaution level. As there is a certain risk for several measures to become counterproductive with changing climate or create trade-offs with other measures, the evaluation of the conflict potential and the precaution level is extremely important.
6. **Applicability**. Most of the measures are region specific and/or their effect is strongly dependent on application conditions. In this section information is given about its potential or factual application limits if available.

Separate chapters are dedicated to **five specific adaptation strategies addressed in the REFRESH Project:**

1. the management of riparian areas to control water temperature by the establishment of woody riparian vegetation along streams and rivers;

2. the management of catchment hydrology to maintain flow in streams, water-level in lakes and regular flooding in wetlands;
3. the re-creation of riparian floodplains to buffer against extreme precipitation events and changes in hydrodynamics, and to reduce nutrient flows and humic substances to water bodies;
4. the management of catchment land-use to reduce diffuse nutrient loading and soil erosion;
5. the management of water abstraction from, and effluent discharge to, surface waters.

Measures contributing to these strategies are highlighted in the review.

1 Introduction

In a recent paper published in Medical Hypotheses, Sleator (2010) compared the metabolic functions of microbes and men describing the latter as a symbiotic superorganism. Going further with this comparison we can find striking similarities between human activities on planet Earth with the microbial (yeast) activity in a fermentation reactor:

- Unlimited access to carbon sources (fossil fuels / sugar) induces a massive pulse of proliferation and reproduction;
- Combustion of carbon (either directly as flame burning or metabolically as wet combustion) releases CO_2 to the environment;
- Intensified (industrial / metabolic) heat release changes the heat balance and, through the positive feedback mechanism, accelerates matter cycles in the system. The thermal equilibrium can also be affected by changes in the external energy source (i.e., the change in the heat flux - heating or cooling) and the thermal insulation of the system (the greenhouse effect / covering of the reactor);
- Metabolic wastes (mining and urban wastes, pollutants from industry and agriculture etc. / alcohol, acetic acid, etc.) will accumulate in the environment where, due to the limited capacity and insufficient recycling, their concentrations increase.
- The system is being transferred to the new state by 1) the exhaustion of carbon source, 2) lethal concentrations of metabolic wastes, or 3) lethal changes in the thermal equilibrium caused by altered external heat flux or modified insulation.

Human and microbial activities differ by

- much greater diversity of processes and species involved in global metabolism compared to the (usually monoclonal) fermentation reactor;
- in human society the proliferation in the sense of social welfare based on energy consumption, and reproduction rate based on decreased mortality, are often spatially separated that creates social tensions and/or migrations;
- the (expected) ability of human society to predict and manage situations to a certain extent and avoid the adverse effects of changes depending on the level of understanding of the causal mechanisms involved, the uncertainty levels of the processes, and cost of the possible measures.

Lower organisms are as well able for adequate sensory behaviour in response to changes in the environment (for example, the taxes as movements of organisms in respect of a directional stimulus or gradient of stimulus intensity). Their adaptation strategy, however, is based on a high turnover rate and a large genotypic plasticity, which allows their fast evolution. Good example of this is the emergence of are drug resistant microbial strains (e.g., Hastings & Watkins, 2005) and strains able to decompose synthetic materials (e.g., Nishida et al., 2000). In human society, where big changes have already occurred or are predicted within a time frame of just a few generations, the microbial adaptation strategies are not viable because the human evolution rate is unable to keep pace with external change. Major mental (science & technology), political and organizational efforts are needed to specify the causal mechanisms behind global change and bring them under control as far as possible (mitigation). The impact of non manageable adverse changes to the society can be softened through various adaptation strategies.

Wilk & Wittgren (2009) divided climate change adaptation strategies into **planned adaptation**, which specifically take climate change and variability into account and **autonomous adaptation**, which goals are not specifically climate related, but that improve resilience to climate change as an additional effect. Because climate change effects on water have a clear and independent manifestation only in few cases until now (e.g. the retreat of glaciers, sea level rise or changes in ice phenology), there is also only a small number of measures that can be qualified as planned adaptation. As examples here could serve the gradual broadening of the North Sea coast with sand nourishments (**M360**), strengthening and raising sea dikes (**M365**) proposed by The Netherlands, and setting norms for increasing water withdrawals for snow making in ski resorts (**M194**) proposed by Austria. The bulk of the adaptation measures belong still to autonomous adaptation where the climate change aspect is often vaguely defined.

The span of measures is wide both by the main purpose (flood management, water scarcity, water quality, biodiversity, CC mitigation), type of intervention (legislative, administrative, financial, educational, hydrotechnical, technological, land use), and especially by the scale of generalization. For instance, under „no regret‟ measures France has listed the implementation of river basin plans while Denmark has proposed the construction of stormwater retention basins (Nixon, 2008). This situation reflects the fact that there is no consistency in using the terms ’measure’, ’action’ or ’strategy’ in the climate change literature and their hierarchic position and linkages with other measures is often obscure.

In overview publications the climate change adaptation measures are commonly grouped by their primary purpose (flood, drought, water quality) while the same measures have often positive effects in all these aspects. For example, restoration of wetlands will suppress the peak flows, create a buffer for alleviating droughts and low flow and will substantially contribute to water quality and biodiversity.

Given the overlapping character and the enormous scale differences among measures, the only manageable format for making a comprehensive overview seems to be of an encyclopaedia type. As this was not feasible in full scale within the person-months available for this task, we selected an approach where the measures are grouped under a limited number of general principles which follow the generally conservative spirit of the environmental sustainability concept. Short articles are given for a selection of measures to characterize the scales and allow a closer insight to possible approaches in different sectors. The full list of measures with relatively scarce information for each single measure is presented in a database format with a detailed categorisation for search purposes.

The need for sustainability arose from the recognition that the profligate, extravagant, and inequitable nature of current patterns of development, when projected into the not-too-distant future, leads to biophysical impossibilities (Goodland, 1995). The resulting goal of environmental sustainability is the unimpaired maintenance of human life-support systems - environmental sink and source capacities. As both human activities and climate change have globally intensified the water cycle (Huntington, 2006) and the mobility of substances (Hesterberg, 1998; Holland & Turekian, 2007), combating of the adverse impacts must be conservative and knowledge based. In this report we grouped the measures by the following three simple principles:

1. **Keep things in place**
2. **Keep things natural**
3. **Be informed and plan your actions**

Each of these three principles are divided into a number of sub-principles, which summarize the essence of the measures aiming at environmental, social and economic sustainability in water management and are illustrated by a number of examples from the list of measures.

As the IPCC report shows, besides a slow change of parameters, climate change is characterized by increased frequency of extreme events. Another most sensitive indicator of climate change is the time shift of seasonal events, i.e. changes in phenology.

If a few degrees change in mean temperature may be not very noticeable (given that the annual amplitude in temperate regions ranges over 50-60 degrees Celsius), the unexpected heat waves and cold spells often have detrimental impact on ecosystems (Gómez & Souissi, 2008) and human mortality rates (Conti et al., 2005). Shifts in the timing of the snowmelt (Bayard et al., 2005) and ice breakup (Goulding et al., 2009) affect the seasonal pattern of runoff, may cause ice jams in rivers, and shifts in phyto- and zooplankton development (Nõges et al, 2010), and fish spawning (Mooij et al., 2008). There are already clear increasing trends in winter runoff and lowering of the spring flood peak in the northern Europe (Saarinen et al., 2010). Large hydrological changes have occurred in watersheds at the permafrost boundary (Wang et al., 2009).

Recent climate projections (Räisänen et al., 2004) indicate an increase of precipitation in Northern Europe and a decrease in Southern Europe. The intensity of single rainfalls, however, is predicted to increase even in regions where the overall amount of precipitation decreases. This will cause flash floods, which may cause great economic damage in densely populated areas. Urbanization as one of the globalization phenomena leads to a rapid growth of impermeable surfaces (buildings, streets and roads, industrial areas, parking places), which further accelerates the runoff from urbanized areas.

Extreme events and seasonality changes represent the major challenges the adaptation measures have to address most urgently. Sea level rise puts several low lying areas in Europe under risk and requires long-term spatial planning and implementation of specific adaptation strategies to guarantee the safety and welfare.

2 Keep things in place

Environmental sustainability or maintenance of life-support systems is a prerequisite for social sustainability (Goodland, 1995). Uncontrolled consumerism in developed nations, marked by increased use of environmental resources, competes with escalating population growth in developing nations as principal threats to environmental sustainability. The impact of any population or nation on environmental sources and sinks is a product of its population, its level of affluence, and the damage done by the technologies that support that affluence (Goodland et al., 1994). Environmental appeals to change consumption behaviour implicitly ask people not merely to change their behaviour but to change their sense of personal identity (Hamilton, 2009). This can be threatening and makes the emergence of a new ecological consciousness more difficult.

Sustainability economics includes the problem of maintaining economic growth, while reducing pollution and/or its impacts, with special attention to the linked problems of energy supply, climate change and – most urgently – fossil fuel consumption. For the sake of progress and economic growth, mankind has accelerated the flow of materials and energy from the environmental sources to environmental sinks. The environmental source and sink capacities are large but finite. Sustainability requires their maintenance rather than exhausting.

The leading role of anthropogenic carbon emissions in the acceleration of climate change has been unequivocally proven by the IPCC (2007), however, the scientific understanding of biophysical linkages of climate change to water cycle and aquatic ecosystems is still weak. There is much uncertainty, and hence an undeniable need for the wide application of the precautionary principle.

2.1 Keep carbon in its present pools

According to the review by Lal (2008), the fluxes among the five global C pools (Fig. 1) are strongly anthropogenically influenced by fossil fuel combustion (>7.5 Pg C y^{-1} during the 2000s) and land use conversion (deforestation) and soil cultivation of about 1.6 Pg C y^{-1}. The total anthropogenic emission of about 9.1 Pg C y^{-1} is balanced by retention of 4.1 Pg C y^{-1} (45%) in the atmosphere, uptake of 2.5 Pg C y^{-1} (27.5%) by ocean, and absorption of 2.5 Pg C y^{-1} (27.5%) by an unidentified terrestrial sink.

The process by which carbon sinks remove carbon dioxide from the atmosphere is known as carbon sequestration. During the 1980s and 1990s, global terrestrial ecosystems took up carbon at a rate of 1–4 Pg y^{-1} offsetting 10–60% of the fossil-fuel emissions (IPCC, 2007; Houghton, 2007). Because growing vegetation absorbs carbon dioxide, the Kyoto Protocol allows countries with large areas of growing forests to issue 'removal units' to recognise the sequestration of carbon. In the Clean Development Mechanism, only afforestation and reforestation (M029) are eligible to produce certified emission reductions in the first commitment period of the Kyoto Protocol until 2012 (LeBlanc, 1999; Olschewski et al., 2005).

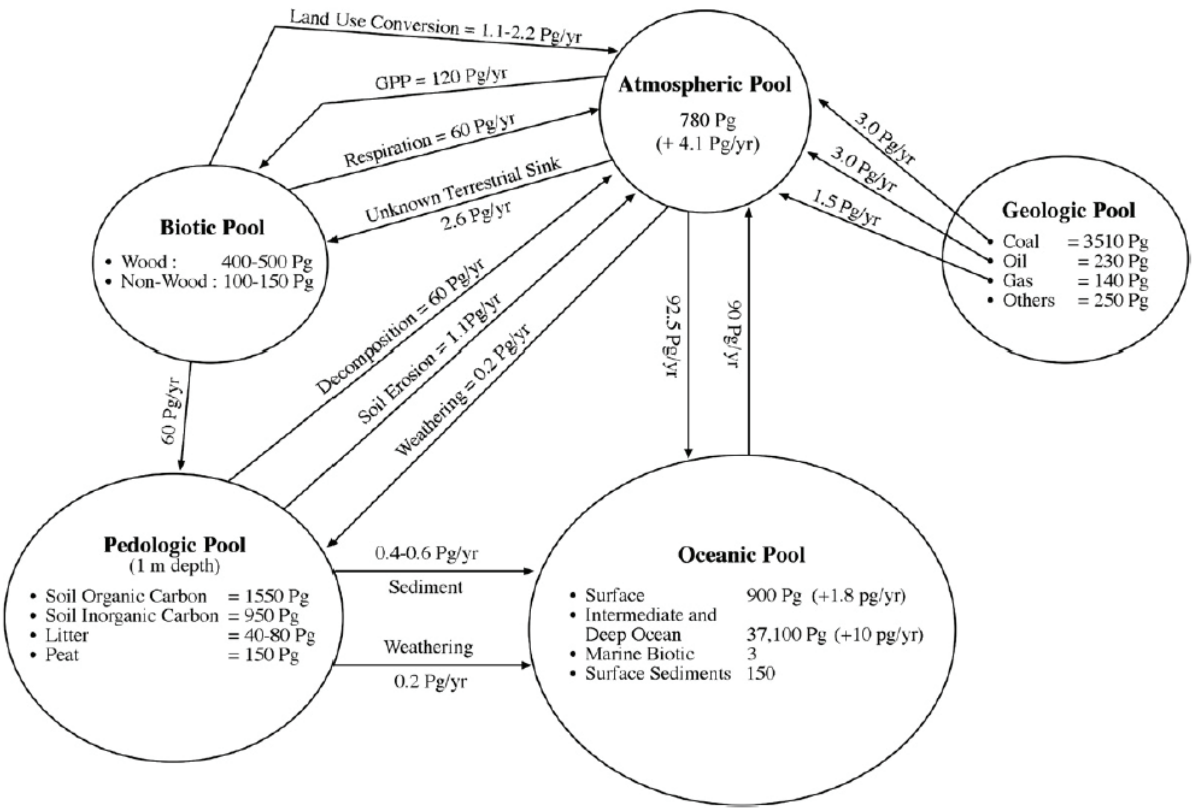


Fig. 1 *Estimates of the global carbon pools and fluxes between them (Lal, 2008)*

Currently, the magnitude of the terrestrial carbon sink is decreasing by expanding land use (House et al., 2003; Hese et al., 2005).

Soils represent a short to long-term carbon storage medium, and contain more carbon than all terrestrial vegetation and the atmosphere combined (Fig. 1). Organic matter tends to accumulate in litter and soils of colder regions such as the boreal forests of North America and the Taiga of Russia. Peatland drainage results in substantial emissions of carbon dioxide and nitrous oxide that should be addressed in a post-2012 climate policy framework. The global figures presented by Parish et al. (2008) show that from the 550 Gigatonnes of peat carbon pools 2 Gigatonnes per year are annually emitted as CO_2 from degraded peatlands (including fires). In sub-tropical and tropical climate conditions, leaf litter and humus are rapidly oxidized and poorly retained due to high temperatures and extensive leaching by rainfall (Powers & Schlesinger, 2002).

At present, agriculture and associated land use changes emit about a quarter of the carbon dioxide (through deforestation and soil organic carbon depletion, machine and fertilizer use), half of the methane (via livestock and rice cultivation), and three-fourths of the nitrous oxide (through fertilizer applications and manure management) annually released into the atmosphere by human activities (Rosenzweig & Tubiello, 2007).

Because freshwater ecosystems cover only a small fraction of the Earth’s surface area, lakes, rivers, and reservoirs have rarely been considered as important components of the carbon cycle at either global or regional scales. Inland aquatic systems are included in global models usually only for the transport of C through the riverine pipe. The review by Cole et al. (2007) on the role of inland waters in the global carbon cycle indicated that the 1.9 Pg C y^{-1} delivered from land to the freshwater exceeds the carbon finally delivered to the ocean by at least a factor of two.

According to this review, lakes, reservoirs, large rivers, floodplains and estuaries are net sources of CO2 to the atmosphere oxidising part of the carbon inflow from the watershed. Lakes contribution on average 0.11 Pg C y^{-1}, reservoirs 0.28 Pg C y^{-1}, the main channels of large rivers 0.23 Pg C y^{-1}, the inundated floodplains of the humid tropics 0.9 Pg C y^{-1}, and estuaries 0.12 Pg C y^{-1}. Ground water contributes a relatively small amount of CO_2 to the atmosphere (about 0.01 Pg C y^{-1} with large uncertainty. It is considered that wetlands still constitute a significant global net sink for CO_2 (Roulet, 2000; Roehm, 2005).

By averaging the published estimates, Cole et al. (2007) reported an annual global storage of 0.05 Pg C y^{-1} for lake sediments that is about 30–60% as much organic C per year as the oceans store, but lakes do this in less than 2% of the area of the sea. According to the recent estimate by Downing et al. (2008), lakes bury even more organic carbon in their sediments than the entire ocean. Lake carbon burial can represent an important part of the total carbon stored in the watershed at the regional scale.

Carbon storage in sediments may be enhanced by eutrophication, reservoir and small pond construction, which slow down the flow rate. As gross primary production (GPP; carbon uptake rate) in lakes is mostly limited by phosphorus, Hanson et al. (2003) suggested that lakes with high total phosphorus (TP) concentrations and low dissolved organic carbon (DOC) concentrations tend to function as net carbon sinks, whereas lakes with low TP and high DOC tend to emit CO_2. Cole et al. (2000) showed that a lake has a net heterotrophic C balance at the mean seasonal chlorophyll a concentration below 20 mg m^{-2} and at GPP less than 140 mmol C m^{-2} day^{-1} or, assuming a 200 day ice-free season, a GPP below 330 gC m^{-2} $year^{-1}$. In their review Andersson and Sobek (2006) showed that the switching from net sink to net source occurred at DOC concentrations higher than 4–6 mg l^{-1}. Smith et al. (2002) and Downing et al. (2008) suggest that also small farm ponds may be quantitatively significant.

Possible CC mitigation measures in water management

Reduction of CO_2 atmospheric loading can be achieved by biological, chemical and technological options through either reducing or sequestering emissions.
Hydropower continues to serve as an important alternative energy source to fossil fuel and nuclear power in many parts of the world and is the cheapest way to generate electricity today. The rise of public awareness of environmental issues of the early 1970s narrowed public acceptance of hydropower as an energy source and reduced significantly its role in the energy matrix in numerous countries (Sternberg, 2008). Measures proposed by the EU Member States regarding hydropower production vary by their scopes implying development of large and micro-scale hydropower capacities (Example 1; **M003**), dam removal

(**M004**), research on future water needs (**M208**), and establishment of rules for the minimum residual flows at hydropower plants (**M278**).

Contemporary hydropower projects and those under construction include environmentally sensitive technical improvements to minimize the project's environmental impact and strike the right balance between the objectives for the water environment and for reducing greenhouse gas emissions. For example, before considering authorising proposed new hydropower schemes, the Scottisch Environment Protection Agency (SEPA) will ensure that:
• all practicable mitigation measures will be taken to minimise the adverse effects of the scheme on the water environment;
• the benefits of the scheme to sustainable development (e.g., reduced emissions of carbon dioxide) outweigh the benefits of preventing deterioration of status;
• the benefits of the scheme cannot be realised by other means representing a significantly better environmental option and not entailing disproportionate cost (SEPA, 2009).

Similarly, growing biofuel crops on arable lands (**M373**) could be a significant alternative to fossil fuels (Falloon & Betts, 2010) that, in addition, could reduce nitrate losses (Powlson et al., 2001) and soil erosion (Börjesson & Berndes, 2006). The biggest concerns, however, are related with increased uses of water, fertilizers and pesticides (Prabhakar & Elder, 2009; de Vries et al., 2010).

A review of mitigation strategies in agriculture (Rosenzweig & Tubiello, 2007) showed that over the next 40 years, „best practice“ and conservation tillage (**M092**) alone could store about 8 GT C in agricultural soils. The „best practice'' agricultural techniques, such as use of catch and cover crops and/or nitrogen fixers in rotation cycles (**M070**; **M072**; **M082**), mulching (**M091**), optimal use of fertilizers (**M073; M080**) and organic amendments; soil water management improvements to irrigation (**M011; M093; M242; M290**) and drainage (**M009; M021**), as well as the conservation tillage evolved as means to enhance sustainability and resilience of agricultural systems to water scarcity rather than with carbon sequestration in mind. Soil carbon stocks can be increased also by converting cropland to grassland or forest to increase soil C sequestration (Ogle et al., 2003; Falloon et al., 2004; Ostle et al, 2009).

Afforestation and reforestation (**M029**, see Example 2) have a number of positive on-site hydrological effects but are also qualified as effective climate change mitigation measures due to carbon sequestration in growing biomass and forest soil. Also peatlands are storehouses of large carbon quantities, thus reducing atmospheric greenhouse gases. However, peatlands remain carbon 'sinks' only as long as they remain in good status. Protection of wetlands (**M115; M132; M307**) and their restoration (**M018; M030-032**, see Examples 3 and 4) can contribute to lowering carbon emissions.

Forest and peat fires release absorbed carbon back into the atmosphere, as does deforestation due to rapidly increased oxidation of soil organic matter. Creation of water retention reservoirs (**M010; M022; M023; M282**) in forested landscapes could supply water for forest fire protection and thus be considered an emission reduction measure especially in the context of the projected increase in the forest fire frequency (Flannigan et al., 2000).

The guidance document on climate change issues in river basin management (CIS, 2009) suggests a 'climate checking' of the planned water management measures as a sensitivity analysis of the proposed measures to evaluate their long-term effectiveness and cost efficiency under changing conditions. This screening provides a good opportunity to assess also the carbon footprint of the measures. The SEPA, for instance, checked all proposed measures regarding the impacts on CO_2 emissions by putting the following questions (SEPA, 2009):

- Will the solutions lead to an increase or decrease in greenhouse gas emissions?
- Will the action help capture carbon in the soil or in vegetation?
- Will the action reduce energy use in the long-term?

Example 1

M003 Development of large and micro-scale hydropower capacities

Description of the problem

Need to increase the proportion of renewable energy sources in the energy budget.

Description of the measure

Hydropower is the most important and widely-used renewable source of energy contributing 19% to the world electricity production[1]. Hydropower produces essentially no carbon dioxide or other harmful emissions, in contrast to burning fossil fuels, and is not a significant contributor to global warming through CO_2.

There are three types of hydropower facilities[2]: impoundment, diversion, and pumped storage. Impoundment facility is the most common type of hydroelectric power plant. An impoundment facility, typically a large hydropower system, uses a dam to store river water in a reservoir. Water released from the reservoir flows through a turbine, spinning it, which in turn activates a generator to produce electricity. The water may be released either to meet changing electricity needs or to maintain a constant reservoir level. A diversion facility channels a portion of a river through a canal or penstock. It may not require the use of a dam. When the demand for electricity is low, a pumped storage facility stores energy by pumping water from a lower reservoir to an upper reservoir. During periods of high electrical demand, the water is released back to the lower reservoir to generate electricity.

Facilities range in size from large power plants (>30 MW) that supply many consumers with electricity to small (100 kW - 30 MW) and micro plants (<100 kW) that individuals operate for their own energy needs or to sell power to utilities.

[1] USGS, Hydroelectric power water use. http://www.usgs.gov/
[2] U.S. Department of Energy, http://www1.eere.energy.gov/windandhydro/hydro_ad.html

Benefit in current climate

Replacement of fossil fuels with renewable energy sources represents a climate change mitigation measure. In addition to clean electricity, impoundment hydropower creates reservoirs that offer a variety of recreational opportunities, notably fishing, swimming, and boating. Most hydropower installations are required to provide some public access to the reservoir to allow the public to take advantage of these opportunities.

Contribution to adaptation to climate change

Hydropower reservoirs can be used for water supply and flood control.

Potential conflicts and problems under changing climate, precaution level

Fish populations can be impacted if fish cannot migrate upstream past impoundment dams to spawning grounds or if they cannot migrate downstream to the ocean. Upstream fish passage can be aided using fish ladders or elevators, or by trapping and hauling the fish upstream by truck. Downstream fish passage is aided by diverting fish from turbine intakes using screens or racks or even underwater lights and sounds, and by maintaining a minimum spill flow past the turbine[2].

Hydropower can impact water quality and flow. Hydropower plants can cause low dissolved oxygen levels in the water, a problem that is harmful to riparian (riverbank) habitats and is addressed using various aeration techniques, which oxygenate the water. Maintaining minimum flows of water downstream of a hydropower installation is also critical for the survival of riparian habitats[2].

Future changes in the occurrence of low flows and droughts may also affect the output of hydroelectric power plants. Of the 40 European countries investigated by Lehner et al. (2001), 14 were indicated as experiencing a future decline of more than 25% in developed hydropower potential.

New hydropower facilities impact the local environment and may compete with other uses for the land. Those alternative uses may be more highly valued than electricity generation. Humans, flora, and fauna may lose their natural habitat. Local cultures and historical sites may be impinged upon. Some older hydropower facilities may have historic value, so renovations of these facilities must also be sensitive to such preservation concerns and to impacts on plant and animal life[2].

Applicability

Presently installed hydropower in Europe totals approximately 179,000 MW (Barnes, 2009). European countries with the largest amounts of hydro include France, Italy, Norway, and Spain. Maintaining and, in many cases, upgrading, this existing infrastructure continues to be an important focus throughout

Europe. Examples of new projects include: Sonna in Norway (270 MW), Glendoe in the United Kingdom (100 MW), and Blanca in Slovenia (42.5 MW).

The emphasis in Western Europe is retrofitting hydro plants with modern equipment, usually upgrading the capacity of the plant. In Eastern Europe, the focus is rehabilitating aging plants that often were allowed to deteriorate during the era of the Soviet Union.

Example 2

M029 Forest restoration

Description of the problem

Many countries of the World including China and several European countries have deforested the majority of their historical forests (Kaplan et al., 2009). Tree removal by logging, forest fire, or wind damage decreases evapotranspiration and increases runoff (Trabucco et al., 2008). Land clearance forces the soil to try to cope with additional water as annual crops and pastures use less water than the deep rooted, native vegetation they replaced. **In humid areas**, changes in the local water balance contribute to waterlogging. **In arid zones**, once forest cover is destroyed, the land may dry and become inhospitable to new tree growth. Together with overgrazing by livestock, and over-harvesting of forest resources it may lead to desertification and the loss of topsoil through erosion.

In Western Australia deforestation for agricultural land has reduced evapotranspiration and lead to increased groundwater recharge and rising groundwater levels. Salts stored in the unsaturated soil zone dissolve and are then precipitated at or very near the surface, or washed into the streams to cause the salinisation both land and streams (Pickering & Owen, 1997).

Description of the measure

Afforestation is planting seeds or trees to make a forest on land which has not been a forest recently, or which has never been a forest. In the UK afforestation may mean legally converting land into a royal forest.

Reforestation is the reestablishment of a forest after removal, for example from a timber harvest. Many countries have experienced centuries of deforestation, and some governments and non-governmental organisations directly engage in programs of afforestation to restore forests and assist in preservation of biodiversity.

The Clean Development Mechanism-Afforestation/Reforestation (CDM-AR) provisions of the Kyoto Protocol allow for carbon sequestration offsets to meet emission reduction obligations for the developed countries, through the purchase of ‘carbon credits’ from afforestation or reforestation projects in developing countries (Trabucco et al., 2008).

The European Union has paid farmers for afforestation since 1990, offering grants to turn farmland back into forest and payments for the management of forest. Between 1993 and 1997, EU afforestation policies made possible the re-forestation of over 5,000 square kilometres of land. A second program, running between 2000 and 2006, afforested in excess of 1000 square kilometres of land (precise statistics not yet available). A third such program began in 2007[3].

Benefit in current climate

On-site hydrological effects of afforestation are mainly positive (Trabucco et al., 2008):

- reduced runoff and erosion,
- improved microclimate,
- increased control over nutrient fluxes,
- decreased sediment loads,
- increased water quality)
- decreased downstream flood risk.
- control stream salinity (van Dijk et al., 2007)

Contribution to adaptation to climate change

Afforestation/reforestation decrease flood risk by increasing on-site evaporation and water retention in the biomass.

Potential conflicts and problems under changing climate, precaution level

Afforestation of upland catchments with fast growing plantations can have significant impact on in situ water use, with consequent impacts on water availability downstream. If converted to forest, about 27% (200 Mha) deemed suitable for CDM-AR prevalent in drier areas would exhibit an 80–100% decrease in runoff. It will become increasingly important to consider implications on local to regional water resources, and how the hydrologic dimension of CDM-AR impacts on issues of sustainability, local communities, and food security (Trabucco et al., 2008).

Applicability

Whether this measure has a positive or negative impact on water resources, water management, soil and land conservation, biodiversity, and/or downstream food security, is highly site specific, and dependent upon climate, soil types, topography, land uses, population densities, existing infrastructures, and tradeoffs with coexisting demands for water (Trabucco et al., 2008).

[3] http://www.e-uropa.osa.pl/-znak-Afforestation

Example 3

M378. Peatland restoration and peat rewetting

Description of the problem

Need to restore aquatic ecosystems as well as the water storage capacity of the landscape. Peatland drainage results in substantial emissions of carbon dioxide and nitrous oxide that should be addressed in a post-2012 climate policy framework. The global figures presented until now (Parish et al., 2008) show that from the 550 Gigatonnes of peat carbon pools two Gigatonnes per year are emitted from degraded peatlands as CO_2 (including fires).

Description of the measure

Raised bogs have been drained and the peat mechanically harvested. On shallower peat, such as the extensive, treeless **blanket bogs** of northern Scotland, the main human intervention is drainage of the peat and the planting of exotic tree species. Regardless of the cause or nature of peatland degradation, the goal of restoration is often to return the degraded site to as near its original state as possible, in terms of both ecological function and habitat for native flora and fauna. **Restoration of the ability to store water is the first priority**. In both types of degraded site, water losses are reduced directly by blocking drains or, more rarely, by installing waterproof membranes along the perimeter. On cutover peatlands, drains are sometimes filled with peat and the bare surface reshaped to create dams or bunds that reduce overland flow. Opportunities for such large-scale engineering are more limited on planted peatlands, because the surface vegetation is still intact. Recent restoration of planted sites in northern Scotland included felling exotic trees and placing them into the ditches intact. The expectation is that branches of the felled trees will act as a climbing frame, enhancing growth of the peat- forming moss, *Sphagnum*, in the ditches (Belyea, 2004).

Benefit in current climate

Contribution to good ecological status, cutting of hydrological extremes.

Contribution to adaptation to climate change

The frequency of flood events is expected to increase as a result of climate change. The measure was marked as win-win measure by Belgium (Nixon, 2008) as increased water storage capacity of the drainage basin helps to alleviate floods and provides water resources to overcome droughts.

Huge reductions of carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions can be attained by rewetting drained peatlands.

Potential conflicts and problems under changing climate, precaution level

A post-2012 framework aiming at peatland rewetting must, however, also address associated methane (CH_4) emissions. The scientific data base for CH_4 emissions from peatland is much larger than that for CO_2 or N_2O. The data show that, once anaerobic conditions are given, the availability of fresh plant material is the major factor in methane production (Couwenberg, 2009).

A recent study carried out in drained peatlands (bogs and minerotrophic mires, located in the Sumava Mountains, Czech Republic, showed that some nutrients (especially nitrogen) and DOC leaching may be expected from drained fens after restoring their water regime (Urbanová et al., 2010).

Applicability

Re-wetting of peatlands is currently a widespread method for restoration of wetland ecosystems which were drained due to intensification of forestry and agriculture (Urbanová et al., 2010).

Example 4

M030-M032 Wetlands and wetland restoration

Description of the problem

Despite their importance as natural habitats and systems buffering the climate change impacts, wetlands are among the most highly threatened ecosystems on the planet. They have suffered continuous degradation and loss. About one half of the world's wetlands have disappeared in the last century[4]. In Europe this value reaches 2/3 and despite conservation measures applied under Ramsar Convention, Natura 2000 network, EU Birds Directive (1979) and Habitats Directive (1992), this trend continues, albeit more slowly (EC, 2007). For the most part, this loss and degradation is caused by

- drainage for agriculture,
- infrastructure developments,
- forestation and malaria control,
- blocking and extraction of the water inflow,
- over-exploitation of groundwater resources,
- building of dams,
- pollution from agricultural and industrial sources.

Approximately 60% of the world's wetlands are peatlands. These areas are sensitive to even the slightest environmental changes. Warmer temperatures, for example, begin to dry out these habitats, resulting in their degradation and, ultimately, complete destruction. Palsa mires, Scandinavian northern mire complexes with permanently frozen peat hummocks are already starting to melt as a consequence of rising temperatures (EC, 2007).

[4] www.wetlands.org

Of particular importance is the role of peatlands as storehouses of large quantities of CO_2, thus reducing atmospheric greenhouse gases. However, peatlands remain carbon 'sinks' only as long as they remain in good status. When damaged, drained or burnt, or when peat is extracted for fuel, peatlands turn from being net carbon sinks to net carbon 'sources'. Therefore, the maintenance of peatlands in good condition is an invaluable asset in the fight against climate change (EC, 2007).

Description of the measure

Wetlands can be more broadly categorised into seven general types (COM, 1995):

- marine and coastal wetlands
- estuaries and deltas
- rivers and floodplains
- lakes
- freshwater marshes
- peatlands
- man-made wetlands, such as canals and reservoirs

The United States Natural Resources Conservation Service[5] has elaborated a number of Conservation Practice Standards regarding different procedures of wetland management:

- **Shallow Water Development and Management** (code 646), i.e. the inundation of lands to provide habitat for fish and/or wildlife.
- **Wildlife Watering Facility** (648) intended to provide watering places for wildlife;
- **Wetland Restoration** (657) intended to rehabilitate a degraded wetland.
- **Wetland Enhancement** (659) intended for modification of an existing wetland where specific attributes are targeted by management objectives
- **Wetland Construction** (656) intended to treat point and non-point sources of water pollution;
- **Wetland Creation** (658) for creating a wetland on a site which historically was not a wetland.

These standards define the purpose, application conditions, environmental criteria, and operation and maintenance procedures for each of these practices.

Benefit in current climate

[5] http://www.nrcs.usda.gov/

Long regarded as wastelands, wetlands are now recognized as important features in the landscape that provide numerous beneficial services for people and for fish and wildlife. Some of these services, or functions, include (US EPA, 2001):

- protecting and improving water quality,
- providing fish and wildlife habitats,
- storing floodwaters, and
- maintaining surface water flow during dry periods.

Contribution to adaptation to climate change

Will contribute if maintains all the functions listed above.

Potential conflicts and problems under changing climate, precaution level

Large uncertainties are included in the possibilities to maintain the necessary hydrological regime and loading of the wetland which determines the efficiency of the system as a site for carbon sequestration (Hefting et al., 2003) and denitrification (Clement et al., 2005; Mulholland et al., 2008).

Applicability

A major motive for drainage of wetland has been the control of malaria. Although it has been eliminated from large areas of Europe, in world terms, it is still a major killer. According to Armstrong (2000), the interaction between the desire to preserve wetland ecosystems and the problems of malaria infestation have not been adequately addressed.

2.2 Keep the water in the catchment by creating retention basins and slowing down the run-off

Interestingly, the need to retain water in the catchment arises both in the case of excess water and when the water resources are scarce, the purposes, however, are totally different. In the case of floods, retaining the water in the upper catchment suppresses the peak flows while during drought periods, the retained water can be used for irrigation and other purposes.

2.2.1 Floods

Large amount of surface water itself poses no risk, (as we can ascertain looking, e.g., the map of Finland) but instead is a valuable natural resource until water bodies remain within their boundaries. The EU Floods directive (Directive, 2007) defines a flood as a temporary covering by water of land not normally covered by water. Throughout Europe different types of floods occur, such as river floods, flash floods, urban floods and floods from the sea in coastal areas. Floods are natural phenomena and their economic damage depends on the exposure of population and infrastructures in the flood zone. The damage caused by flood events may also vary across the countries and regions. The larger the disaster and the smaller the economy, the more significant is the impact (Hansson et al., 2009). The disaster losses have grown mostly due to investments made in the flood risk areas (Benson & Clay, 2004).

In the period 1950–2006, 40% of the flood-related casualties occurred in Europe due to flash floods (Barredo, 2007) and the potential is increasing in many regions due to the social and economic development bringing pressure on land use. On the other hand, the intensifying hydrological cycle due to climate warming (Huntington, 2006) brins about increasing heavy precipitation (Groisman et al., 2004, 2005). As a consequence, the flash flood hazard is expected to increase.

Marchi et al. (2010) who analysed 25 extreme flash floods across Europe found a peculiar seasonality effect on flash flood occurrence, with events in the Mediterranean and Alpine–Mediterranean regions mostly occurring in autumn, whereas events in the inland Continental region commonly occur in summer, revealing different climatic forcing. Sivapalan et al. (2005) showed that climate variability affects flood generation directly through the variability of storm characteristics, and indirectly through the seasonality of rainfall and evapotranspiration that modify the antecedent catchment conditions for individual storm events. Physiographic factors may affect flash flood occurrence by combination of orographic effects augmenting precipitation, and topographic relief promoting rapid concentration of streamflow (O'Connor & Costa, 2004).

Flood management measures

Within the European Union, the objectives regarding the management of flood risks should be determined by the Member States themselves and should be based on local and regional circumstances. The EU Floods directive (Directive, 2007) requests Member States to develop flood risk management plans, which should cover the entire catchment area of watercourses and promote the co-ordinated development, management and conservation of actions regarding water, land and related resources. Such a holistic approach is based on multilateral and even multinational co-operation, including interdisciplinary planning for the whole catchment areas.

Member States are requested to include provision for floods and droughts to the second and third River Basin Management Planning (RBMP) cycles (CIS, 2009). For floods these provisions must include reliable technologies for disaster prevention, early warnings, and mitigation (Hanson et al., 2008).

Among various management measures that can reduce flood damage, there is a growing interest in unconventional methods, involving the transfer of some of the surplus water into areas less liable to flood damage and situated upstream of the zones to be protected (Pivot et al., 2002). A variety of specific flood protection measures and measures contributing to the alleviating of flood risk were included in the database. The strategy for floodplain restoration (**M019**; Example 5) is exemplified on the basis of the spatial planning measures applied in the "Room for the River" programme in The Netherlands (Spatial, 2006). Examples 6-11 include the establishment of retention areas (**M017; M022; M023; M027**), stormwater management (**M020**) and systems of Sustainable Urban Drainage (SUDs; **M005**).

Example 5

M019 Floodplain restoration

Description of the problem

With the land behind the river embankments is becoming more heavily used and populated, the rivers are increasingly forced to remain within their banks. In case of increasing affluence, water level rises quickly and unpredictably and may cause devastating floods in populated areas.

Description of the measure

Floodplain restoration is the process of fully or partially restoring a river's floodplain to its original conditions before having been affected by the construction of levees (dikes) and the draining of marshes. The objectives of restoring floodplains include

- the reduction of the incidence of floods,
- the provision of habitat for aquatic species,
- the improvement of water quality and
- the increased recharge of groundwater.

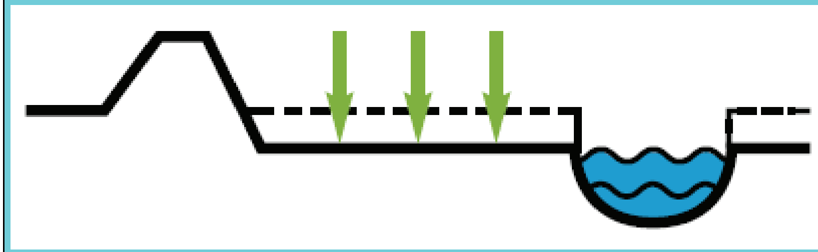
In Europe very few schemes for restoring functional floodplains have been put in practice so far, despite a surge of interest in the topic among policy and research circles. One of the drivers for floodplain restoration is the EU Water Framework Directive. Early floodplain restoration schemes were undertaken in the mid-1990s in the Rheinvorland-Süd on the Upper Rhine, the Bourret on the Garonne, and the Long Eau River project in England. Ongoing schemes include the Spatial Planning Key Decision (SPKD) “Room for the River” in The Netherlands (Spatial, 2006), Lenzen on the Elbe, La Basse on the Seine and the Parrett Catchment Project in England (Moss & Monstadt, 2008). On the Elbe River near Lenzen (Brandenburg) 420 hectares of floodplain were restored in order to prevent a recurrence of the Elbe floods of 2002. A total of 20 floodplain restoration projects on the Elbe River were envisaged after the 2002 floods, but only two have been implemented as of 2009 according to the environmental group BUND (IKSE, 2009).

In the US, examples of floodplain restoration can be found in the catchment area of the Chesapeake Bay in Maryland[6], in the Emiquon Preserve on the Illinois River[7], in Charlotte, North Carolina[8] and along the Baraboo River in Wisconsin[9].

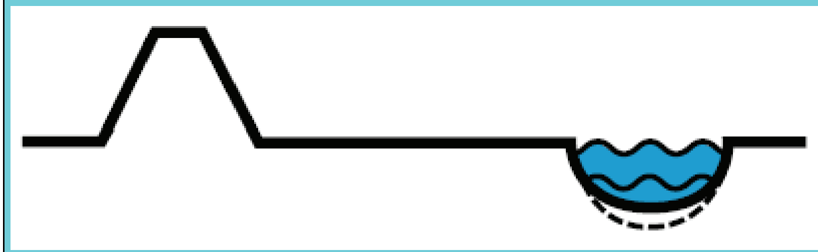
The spatial planning measures applied in The Netherlands include (Spatial, 2006):

[6] http://www.landstudies.com/legacysediments.html
[7] http://www.nature.org/wherewework/northamerica/states/illinois/press/emiquon.html
[8] http://www.charmeck.org/Departments/StormWater/Storm+Water+Professionals/Project+Floodplain+Restoration.htm
[9] http://sandcounty.net/programs/pioneers/floodplain_rehab/

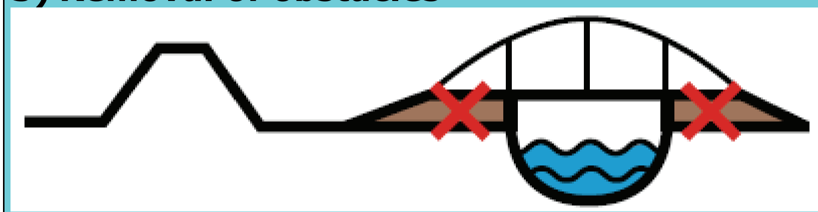
1) Deepening of the forelands



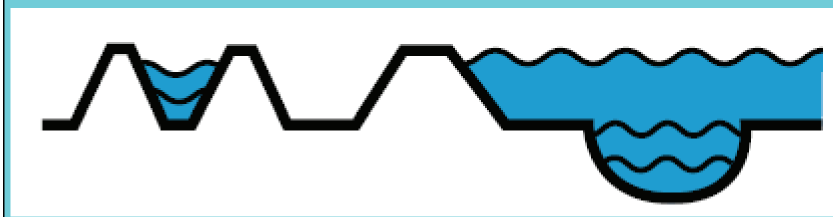
2) Enlarging of summer beds



3) Removal of obstacles



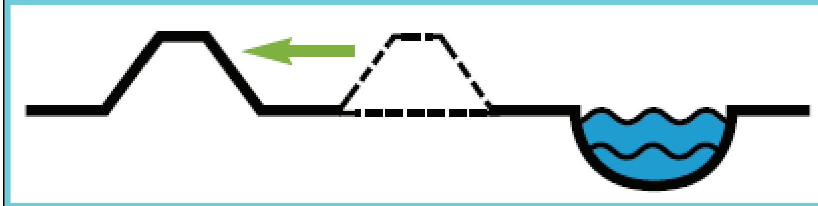
4) Creating of flood channels



5) Lowering of groynes

6) Strengthening of dikes

7) Displacement of dikes or depoldering



Benefit in current climate

The measures reduce the incidence and impact of floods, provide habitat for aquatic species, improve water quality and increase the recharge of groundwater.

Contribution to adaptation to climate change

Extremely high river discharges will occur more frequently in the future. The measures will help to cope with greater volumes of water in a safe manner.

Potential conflicts and problems under changing climate, precaution level

An environmental impact assessment is required for many of the measures. As floodplain restoration means large changes in the landscape and everyday life of the inhabitants, there is need to incorporate public perception into river restoration projects and the potential for project initiators to form strategic alliances with local residents to promote ecological restoration in combination with river restoration.

Increased flooding upstream in the 'flood expansion areas' may affect the agriculture causing partial or total destruction of crops and resulting in serious losses for the farmers. Additional flooding could therefore lead to vigorous opposition from farmers to such management projects (Pivot et al., 2002).

Norway has pointed out (Nixon, 2008) that several flood protection measures may in some cases reduce drainage capacity of rivers.

Applicability

The set of feasible and affordable measures depends on local conditions and needs.

Example 6

M022 Modification of the excess water drainage system (more local retention or reservoirs)

Description of the problem

In many catchments excess drainage water that can be considered as a potential source of water for irrigation, recharging of groundwater or for creating aquatic habitats is unproductively discharged out of the catchment area. Occurrence of drought and excess water in the same area and in the same year is characteristic in many lowland areas e.g. in Hungary (Nixon, 2008). Due to short retention time of water in the catchment, nutrients are not retained but carried into recipient water bodies (lakes, estuaries, coastal areas).

Description of the measure

Creation of small retention ponds, impoundments and reservoirs within catchments

The highest relative increase of water retention in ponds and in adjoining areas could be obtained in midfield and farmstead ponds with the smallest average area. However, for the smallest water bodies with surface area less than 0.5 ha, the potential relative increase of groundwater retention is higher than the retention increase in the pond itself. This means that the smaller the pond and the smaller value of current water body retention, the bigger relative increase of groundwater retention in the areas adjacent to pond in relation to the increase of the water level in pond (Juszczak et al., 2007).

Benefit in current climate

Small ponds and retention reservoirs could serve for

- irrigation
- detention of flood water
- retention of nutrients
- recharging of groundwater
- creating aquatic habitats
- supplying animals with water
- forest fire protection

Contribution to adaptation to climate change

In more severe climatic conditions the stabilizing role and the importance of services offered by small water bodies within the catchment will increase.

Potential conflicts and problems under changing climate, precaution level

Accumulation of high quantities of agrochemicals in ponds in areas of intensive agriculture may impair many of the uses of these water bodies. Increased flooding upstream in the ‘flood expansion areas’ may affect the agriculture causing partial or total destruction of crops and resulting in serious losses for the farmers (Pivot et al., 2002).

Applicability

Where apropriate. No contraindications reported.

Example 7

M017 Storm water detention basins

Description of the problem[10]

As land is developed, the amount of impervious surface increases and the amount of vegetation decreases. Impervious surfaces are surfaces such as asphalt, concrete and rooftops that do not allow water to soak into the ground, which increases both the volume of water in the washes and the peak flow.

Description of the measure

A detention basin is a stormwater management facility installed on, or adjacent to, tributaries of rivers, streams, lakes or bays that is designed to protect against flooding and, in some cases, downstream erosion by storing water for a limited period of a time. These basins are also called "dry ponds", "holding ponds" or "dry detention basins" if no permanent pool of water exists. Some detention ponds are also "wet ponds" in that they are designed to permanently retain some volume of water at all times. In its basic form a detention basin is used to manage water quantity while having a limited effectiveness in protecting water quality[11].

Incoming water begins to fill the storage space within the basin during the first phase of a storm. The basins are equipped with an outlet structure that releases the water more slowly than it comes in. As the storm progresses, the outlet structure causes the water to back up within the basin. The more slowly it is released, the larger (and more expensive) the basin has to be. The designer has to balance these two components of the design to get the most efficient facility possible.

Benefit in current climate

Detention basins help the community in several ways:

1. reduce the danger of downstream flooding by reducing the peak flows. Detention basins are storm water best management practices (BMPs) that provide general flood protection and can also control extreme floods such as a 1 in 100-year storm event.
2. By increasing the retention time of water in the basin, thereby giving it a greater chance to infiltrate back into the ground and recharge the aquifer.
3. By reducing biochemical oxygen demand (BOD) emission to rivers (Nixon, 2008).

A variant basin design called an extended detention (ED) dry basin can limit downstream erosion and control some pollutants such as suspended solids. While basic detention ponds are designed to empty within 6 to 12 hours after a storm,

[10] http://www.sierravistawater.com/stormwtr_basins.htm

[11] http://en.wikipedia.org/wiki/Detention_basin

ED basins lengthen the storage time to 24 or 48 hours resulting in improved water quality because settling of suspended solids.

Contribution to adaptation to climate change

Alleviating floods.

Potential conflicts and problems under changing climate, precaution level

As pointed out by Denmark in the questionnaire on adaptation measures (Nixon, 2008), storm water detention basins alleviate floods and reduce BOD emission to rivers being a no regret measure.

Accumulation of pollutants in ponds in urban areas may impair many of the uses of these water bodies.

Applicability

Where apropriate. No contraindications reported.

Example 8

M023 Construct small reservoirs in hilly regions (HU)

Description of the problem

Hungary is vulnerable to any climatic changes brought about by global warming. There is increasing temperature and decreasing rainfall trends. Over the last several years, average summer temperature increased by 1°C (Makra et al., 2005). The average precipitation on the Great Plain of Hungary is not enough for intensive cultivation, with evaporation consuming too much water. Because of these reasons natural water deficiency occurs regularly every year and resources have to be substituted by man-made means (ICPDR, 2008).

The frequency of floods tripled during the last decade. The continental climate of the Tisza watershed causes a more extreme river behavioral pattern than that of the Danube. Hungary's location is unique as it is in a basin surrounded by mountains that are located in other countries. This means that national practices of Austria, Slovakia, the Ukraine and Romania affect the Hungarian environment (Turnock, 2002).

Hungarian National Climate Change Strategy 2008-2025[12] includes measures for

- flood protection,
- remediation,
- drinking water resource protection

Description of the measure

[12] www.kvvm.hu/cimg/documents/nes080214.pdf

Within the ICPDR Sub-Basin Level Flood Action Plan for Tisza River Basin, Ukraine has planned to construct 42 dry flood retention reservoirs in mountainous part of Tisza basin (ICPDR, 2009).

Benefit in current climate

Harvesting and storing of rain and meltwater into water reservoirs is a possible and potential solution for flood prevention and revitalisation of hilly areas, supplying drinking and irrigation water and water for forest fire protection.

Contribution to adaptation to climate change

Decrease of flood peaks and supplying water for forest fire protection in the case of increasing extremes.

Potential conflicts and problems under changing climate, precaution level

In karst areas widely spread in Hungary, Slovakia, Slovenia, and Italy, agricultural activity is a possible source of pollution of drinking water. Especially high nitrogen surpluses can be caused by high animal density per ha. The stocking rate over 2.1 LU/ha can cause net-balance surplus over 100 kg/ha; in this case organic fertilization can be considered a serious non-point pollution source (Maticic, 1999). The only way of harvesting rainfall in karstic areas, is by construction of ponds sealed with suitable plastic or other material (Maticic, 2004).

Applicability

Needs careful environmental impact assessment regarding local hydrogeological conditions.

Example 9

M025 Planned inundations of protected (former natural) floodplains (HU)

Description of the problem

The level of flood exposure in terms of the ratio of flood plains is the highest in Hungary of all European countries and is comparable with the situation in the Netherlands alone (ICPDR, 2009). The hydrology of Hungary can be characterised by its basin-like situation where more than 95% of the outgoing discharge comes from beyond the country borders and less, than 5% has local origin (Mezosi, 2004). High floods may occur on the Tisza River in any season of the year, but the nature of the floods and their determinant circumstances are very variant and unpredictable. The most effective solution would be a well-designed flood protective system on the upper reaches of the rivers as well. For mitigation of this very exposed situation there are strong efforts also in Hungary aimed at decreasing the flood damages.

Description of the measure

One aim of the Vásárhelyi Plan (2004)[13] was creating a storage-drainage system on the Hungarian flood plains for temporal retardation of 1,500 Mm^3 water in order to achieve gradual reduction of flood peaks along the river.

The following measures were foreseen:

1. Replacement of flood levees
2. Rising of dikes
3. Dredging the flood channel
4. Changing river training concepts
5. Dredging the main channel
6. Revitalisation of wetlands
    a. Supply of refreshing water from the river
    b. Elimination of local sources of pollution
    c. Dredging of bottom, deepening the oxbow lakes
7. Removal of buildings
8. Changing landuses
9. Removal of summer dikes
10. Construction of flood retention reservoirs.

The planning step included (Bódis, 2007):

- multiphase evaluation for selection of the most feasible areas,
- description of the planned reservoirs, their borders, drafting the necessary earthworks and new dams, mapping the location of natural elevated river banks,
- description of the current land use and ownership of the selected properties,
- calculation of the capacity of planned emergency reservoirs and feasibility-analysis.

Taking into account the significance of the Program, it has some drawbacks (ICPDR, 2009):

- it focuses mainly on construction measures,
- it **does not reflect climate change**
- it does not fully fit with requirements of EU Flood Directive 2007, although it reflects some of them (development of flood hazard and flood risk maps).

After joining the European Union in May 2004 the programme was redesigned to match the new financing instruments of the EU. This new programme includes (ICPDR, 2009):

- strengthening of the existing dykes to meet the present design standard (1:100 year flood with 1 m freeboard),
- restoring the flood conveyance capacity of the flood way by opening a 300-600 m wide "hydraulic corridor" (an area with less hydraulic resistance),
- relocating dykes to remove bottlenecks and building flood retention reservoirs to provide extra safety against flood larger than the design flood.

By November 2009 two of the first six flood retention reservoirs (Cigánd and Tiszaroff Flood Retention Reservoirs) and the dyke relocation at Rákóczifalva were completed.

[13] The new Vásárhelyi Plan: http://www.vizugy.hu/vtt/altalanos_english.pdf

Benefit in current climate (Vásárhelyi Plan)

- Flood protection
- Some limited beneficial use of the flood flows
- Better water supply to the background areas
- Lower loss sensitivity and higher profitability of changed land uses
- Support to biodiversity
- Alleviation of flood erosion risk
- Possibility to convert some reservoirs to fish farming

Contribution to adaptation to climate change

Decrease of flood peak in the case of increasing extremes.

Potential conflicts and problems under changing climate, precaution level

The main risk is related to the high uncertainty of climate change projections. Underdimensioned flood protection facilities in combination with growing vulnerability of the modern society may cause big economic losses whereas overdimensioned facilities present wasting of resources.
Norway has pointed out (Nixon, 2008) that several flood protection measures may in some cases reduce drainage capacity of rivers.

Applicability

Depending on the trade-offs between flood risk rate and the development level within the floodplain area.

Example 10

M020 Stormwater management

Description of the problem

Stormwater is a term used to describe water that originates from precipitation events. Stormwater that does not soak into the ground becomes surface runoff, which either flows directly into surface water bodies or is channeled into storm sewers, which eventually discharge to surface waters. In urban environments where **impervious surfaces** (parking lots, roads, buildings, compacted soil) do not allow rain to infiltrate into the ground, more runoff is generated than in the undeveloped condition. This additional runoff can erode watercourses (streams and rivers) as well as cause flooding when the stormwater collection system is overwhelmed by the additional flow. Because the water is flushed out of the watershed during the storm event, little infiltrates the soil, replenishes groundwater, or supplies stream baseflow in dry weather (Schueler, 2000). Stormwater is of concern for two main issues: one related to the **volume and timing of runoff** water (flood control and water supplies) and the other related to potential contaminants that the water is carrying, i.e. **water pollution**.

Description of the measure

Stormwater management means managing the quantity and quality of stormwater. The terms **Best Management Practice (BMP)** (WSDE, 2005) or

Sustainable Urban Drainage (SUD) (see **M005**, Example 11) is used to refer to both structural or engineered control devices and systems (e.g. detention ponds) to treat polluted stormwater, as well as operational or procedural practices. Many design guidelines for such Urban drainage devices are now available (e.g., CIRIA, 2000; NZWERF, 2004; Ballard et al., 2007). Zoppou (2001) and Elliott & Trowsdale (2007) provided an overview of stormwater modelling approaches and described the common mathematical methods for flow routing and contaminant generation and transport calculations.

The types of BMPs include **source control, treatment,** and **flow control measures** (WSDE, 2005).

Source control BMPs prevent pollution, or other adverse effects of stormwater, from occurring. Source control BMPs are classified as
operational or structural. Examples of source control BMPs include various methods as using

- mulches and covers on disturbed soil,
- putting roofs over outside storage areas, and
- berming areas to prevent stormwater run-on and pollutant runoff.

It is generally more cost effective to use source controls to prevent pollutants from entering runoff, than to treat runoff to remove pollutants. However, since source controls cannot prevent all impacts, some combination of measures will always be needed.

Treatment BMPs include facilities that remove pollutants by simple gravity settling of particulate pollutants, filtration, biological uptake, and soil adsorption. Treatment BMPs can accomplish significant levels of pollutant load reductions if properly designed and maintained.

Flow control BMPs typically control the rate, frequency, and flow duration of stormwater surface runoff. The need to provide flow control BMPs depends on whether a development site discharges to a stream system or wetland, either directly or indirectly. Stream channel erosion control can be accomplished by BMPs that detain runoff flows and also by those which physically stabilize eroding streambanks. Both types of measures may be necessary in urban watersheds.

Benefit in current climate

Proper management of stormwater prevents urban floodings and diminishes pollution of runoff water.

Contribution to adaptation to climate change

With the projected increase of precipitation amount and intensity, the importance of proper functioning of stormwater systems will increase.

Potential conflicts and problems under changing climate, precaution level

Altered runoff predicted by climate change has the potential to increase the volume of stormwater that can contribute to drainage and flooding problems. Often, combined sewers can not handle the volume of runoff, resulting in combined sewer overflows and causing water pollution problems in nearby water bodies.

It remains to be seen whether the available continuous runoff models are sufficiently accurate to determine successful flow management strategies. Even if the modeling approaches are sufficient, it will be a challenge to simulate predevelopment hydrology after significant development has occurred.

Applicability

Universal applicability in urban areas. Importance increases with the amount of precipitation and with the proportion of impermeable surfaces.

Example 11

M005 Sustainable Urban Drainage Systems (SUDS)[14]

Description of the problem

Built-up areas need to be drained to remove surface water. Traditionally this has been done using underground pipe systems designed for quantity, to prevent flooding locally by conveying the water away as quickly as possible. The alteration of natural flow patterns can lead to problems elsewhere in the catchment. Water quality issues have become increasingly important, due to pollutants from urban areas being washed into rivers or the groundwater. Once polluted, groundwater is extremely difficult to clean up. Conventional drainage systems cannot easily control poor runoff quality and may contribute to the problem. The amenity aspects, such as water resources, community facilities, landscaping potential and provision of varied wildlife habitats have largely been ignored. Conventional drainage systems are not designed with these wider considerations in mind. Continuing to drain built up areas with limited objectives and ignoring wider issues is not a sustainable long-term option causing an impact on the terrestrial and aquatic environments.

Description of the measure

Surface water drainage methods that take account of quantity, quality and amenity issues are collectively referred to as **Sustainable Urban Drainage Systems (SUDS)** but alternative acronyms are WSUD (water sensitive urban design), LIUDD (low impact urban design and development, a term used in New Zealand), and LID (low impact development) devices (Elliott & Trowsdale, 2007). These systems are more sustainable than conventional drainage methods because they:

- manage runoff flow rates reducing the impact of urbanisation on flooding,
- protect or enhance water quality,
- are sympathetic to the environmental setting and the needs of the local community,
- provide a habitat for wildlife in urban watercourses,
- encourage natural groundwater recharge (where appropriate).

They do this by:

[14] http://www.ciria.org.uk/suds/background.htm

- dealing with runoff close to where the rain falls,
- managing potential pollution at its source now and in the future
- protecting water resources from point pollution (such as accidental spills) and diffuse sources.

SUDS are made up of one or more structures built to manage surface water runoff. They are used in conjunction with good management of the site, to prevent flooding and pollution. There are four general methods of control:

- Prevention
- Filter strips and swales
- Permeable surfaces and filter drains
- Infiltration devices
- Basins and ponds

Sediment basins and traps should be installed before any major site grading takes place. Additional sediment traps and silt fences should be installed as grading takes place to keep sediment contained on site (Ballard et al., 2007).

Benefit in current climate
UK has indicated sediment traps and SUDS as measures that reduce nutrient input and flood risk (Nixon, 2008). They may also allow new development in areas where existing sewerage systems are close to full capacity, thereby enabling development within existing urban areas.

Contribution to adaptation to climate change
Dealing with runoff close to where the rain falls, SUDS reduce the impact of urbanisation on flooding. Attenuation of peak flows decreases soil erosion and landslide risk and contributes to the quality of the runoff water.

Potential conflicts and problems under changing climate, precaution level
UK has indicated at potential risks that SUDS may be not designed to sufficient capacity to cope with future rainfall (Nixon, 2008). The SUDS design philosophy, however, unlike traditional systems, is to use a train of management methods. For example, once the soak away has reached its capacity, the overland flow can be stored in a pond or wetland or underground storage. Flooding, should it occur, can also be managed to reduce impact, for example careful planning and design can ensure that areas such as playing fields should be flooded before roads and that houses are positioned so they are less likely to be inundated.

Applicability
Virtually unlimited in most urban areas. Wide selection of various techniques allows application of SUDS in areas which have little or no infiltration, in contaminated areas and and areas where space is limited.

2.2.2 Water scarcity and drought

Most of the flood water retention basins can contribute to protection against droughts as the excess drainage water collected in one season can be considered as a potential source of water for irrigation or recharging of groundwater if a drought period follows. Numerous retention basins will alleviate the impacts of low flow periods in rivers and create additional aquatic habitats. Rainwater harvesting (**M306**) constitutes a potential source of drinking water and, if properly managed, could reduce water and food crisis in several developing countries suffering from water shortage (Helmreich & Horn, 2009). In Egypt, for example, rainwater harvesting is an alternative to the more expensive desalination of brackish groundwater (**M374**; Allam et al., 2003). Even in Ireland, where abundant water resources are available because of plenty of rainfall, a future water shortage is anticipated, especially in urban areas. The use of domestic rainwater harvesting and greywater treatment systems has the potential to supply nearly 94% of domestic water in Irish households helping to achieve significant water savings (Li et al., 2010).

Several methods such as building terraces and contour barriers (**M139**), backfilling of ravines (**M140**), and stormwater management measures (**M020**; Example 10) can be used to slow down the runoff of rainfall waters. Additional storage capacities can be created turning ravines into ponds (**M372**), fish ponds rehabilitation, refurbishment, and sludge removal (**M170**), or by sediment removal from shallow lakes (**M379**).

In drought prone areas, water saving systems achieve high importance in all economic sectors where they should increase the efficiency of water use and solve quantitative unbalance. On a general level, the measures include water metering (**M096; M306**), permits for water abstraction (**M247-251; M258; M259**), water pricing (**M044**), water distribution systems (**M090; M246**), safety water technologies (**M089**), and new technologies of water recovery, such as recycling and infiltration (**M084**).

Entire strategies have been elaborated for agriculture to reduce its vulnerability to water scarcity and drought including drought tolerance improvement in crop plants (Cattivelli et al., 2008), efficient irrigation (Sanches et al., 2009; **M093**) and drainage systems (**M009; M021**; Examples 12 and 13), and agrotechnical measures to protect soil moisture, such as mulching (**M091**), cover- and catch crops (**M070; M072; M082**), organic farming (**M066**), deep ploughing (**M068**), and conservation tillage (**M092**).

Water scarcity problems have called into being a number of water reuse projects. The WSP (Water Safety Plans) session at the 5th World Water Forum in Istanbul, 2009 under the title "MUFS[15] for more MDGs[16] per drop; how to make it happen?" recommended that wastewater should be recognised as an important resource in national water policies, IWRM[17] strategies and national and local water budgets.

Example 12

M009 Decrease groundwater drainage

Description of the problem

[15] Multiple Use and Functions of Water Services
[16] Millennium Development Goals
[17] Integrated Water Resource Management.

Watercourses are important for the interaction between surface water and groundwater. The current dredge and fill practices in locating canals along the periphery of wetlands are transforming natural basins that originally had primarily slower subsurface drainage to ones that discharge larger quantities of water faster, via a surface drainage system (Wang & Overman, 1981). Overdrainage, i.e. excess lowering of the groundwater table may influence the yield of major crops in drought conditions (Khan et al., 2003).

Description of the measure

The measures foreseen in order to raise groundwater levels and thus benefit certain ecosystems are to raise the water levels of the main watercourses and to raise the beds of the smaller watercourses by infilling (Querner & van Lanen, 2001).

Benefit in current climate

As indicated by Hungary, provides better quantitative conditions of the groundwater body (in some areas conversion from arable land to grassland is necessary).

Contribution to adaptation to climate change

Increase of recharge. Maintaining shallow groundwater level; water supply of vegetation by capillary water.

Potential conflicts and problems under changing climate, precaution level

Hungary has indicated it as a win-win measure (Nixon, 2008). In salinisation prone areas of irrigated agriculture, shallow groundwater must remain below the crop root zone with low concentrations of salt. Shallow groundwater can cause progressive salinization of the crop root zone (Drainage Management, 1998).

Applicability (Ayars et al., 2006)

Controlled drainage has been practiced in humid areas for a long time. In arid regions, controlled drainage is the next logical step towards improving water management in irrigated agriculture and reducing the environmental impacts of subsurface drainage flow. The suggested changes include reducing the installation depth of laterals, accounting for crop water use from shallow ground water in the design, and relaxing the mid-point water depth requirement. Active control of drainage systems in arid irrigated regions is a developing concept that is currently being evaluated around the world. Research in the U.S. and Australia has demonstrated that water tables in irrigated areas can be effectively controlled with various types of structures. Control has resulted in reduced volumes of drainage water and total salt loads discharged. Salt accumulation in the root zone is a consideration in adopting controlled drainage, but other research has demonstrated that it is possible to manage salt accumulation through careful water management.

Example 13

M021 Controlled drainage (CD)

Description of the problem

On flat, poorly drained soils, intensive drainage is necessary to facilitate seedbed preparation and planting in order to minimize plant stress and subsequent yield reduction resulting from poor soil aeration that accompanies waterlogging. The drainage intensity required for agricultural production is not the same in all years or all periods of the year. While **wetness is the major concern**, weather conditions vary such that crops periodically suffer from **drought stress** that may substantially reduce yields in some years. Intensive drainage systems, necessary to provide trafficability during extreme wet periods often, remove more water than necessary during drier periods, leading to temporary overdrainage (Doty et al., 1986). With uncontrolled drainage large amounts of nitrates are lost from fields and carried to surface water bodies.

Drainage of **acid sulfate soils** causes specific problems. Acid sulfate soils are naturally occurring soils, sediments or organic substrates (e.g. peat) that are formed under waterlogged conditions. These soils contain iron sulfide minerals (predominantly as the mineral pyrite) or their oxidation products. Acid sulfate soils are widespread around coastal regions, and are also locally associated with freshwater wetlands and saline sulfate-rich groundwater in some agricultural areas. In Australia, coastal acid sulfate soils occupy an estimated 40,000 km^2 (Fitzpatrick et al., 2003), on the coastal plains of Finland there are approximately 3,000 km^2 of acid sulfate soils developed as a result of intensive agricultural drainage of waterlogged sulfide-bearing sediments (Åström et al., 2007). In an undisturbed state below the water table, acid sulfate soils are benign. However if the soils are drained, excavated or exposed to air by a lowering of the water table, the sulfides will react with oxygen to form sulfuric acid (Åström et al., 2007).

Release of this sulfuric acid from the soil can in turn release iron, aluminium, and other heavy metals (particularly arsenic) within the soil. Once mobilized in this way, the acid and metals can create a variety of adverse impacts: killing vegetation, seeping into and acidifying groundwater and water bodies, killing fish and other aquatic organisms, and degrading concrete and steel structures to the point of failure.

Description of the measure (Wesström et al., 2001)

CD makes it possible to vary the drainage intensity with the variation in drainage requirement during season by controlling the height of a riser (Fig. 2) in the drain outlet and thus to a certain degree control the amount of outflow of solutes via the drainage system. During periods with low drainage demand, the riser in the drain outlet can be raised and the groundwater level in field will rise up to the level of the riser before the discharge takes place.

The successful management of CD systems rests on two important objectives. The first is achieving optimum production efficiency and maximum nutrient utilization by the crop; the second is attaining maximum water quality benefits.

Fig. 2 *Controlled drainage system showing flashboard riser* (http://www.soil.ncsu.edu/)

Benefit in current climate

First, CD reduces the volume of drainage water leaving a field from 20–30% on average; however, outflow varies widely depending on soil type, rainfall, type of drainage system and management intensity. During dry years, CD may totally eliminate outflow. In wet years, control may have little or no effect on total outflow.

Second, CD provides a higher field water table level which promotes denitrification within the soil profile. In some cases, nitrate-nitrogen concentrations have been 10–20% lower in outflow from controlled systems compared to uncontrolled, free-draining systems. The combined effect of reduced flow and reduced nitrate concentration results in the overall 45% reduction in nitrogen mass transport at the field edge.

CD has also been documented to reduce phosphorus transport by roughly 35%[18].

The CD system if applied on acid sulfate soils, should increase groundwater levels and reduce the sulfide oxidation and the load of discharged acidity and toxic metals into adjacent streams.

Contribution to adaptation to climate change

It is a flexible multipurpose measure especially beneficial for improving water quality and alleviating the impact of droughts to agriculture.

Potential conflicts and problems under changing climate, precaution level

The mechanisms governing the hydrology and loss of pollutants from artificially drained soils are complex and vary with conditions prior to drainage improvements and other factors: land use, management practices, soils, site conditions, and climate. Increasing drainage intensity on lands already in agricultural production may have positive, as well as negative, impacts on hydrology and water quality. For example, increasing the intensity of subsurface

[18] Controlled drainage: What is it and how does it work? http://www.soil.ncsu.edu/publications/BMPs/drainage.html

drainage generally reduces loss of phosphorus and organic nitrogen, whereas it increases loss of nitrate-nitrogen and soluble salts. Conversely, increasing surface drainage intensity tends to increase phosphorus loss and reduce nitrate-nitrogen outflows. Improved drainage is required on many irrigated, arid lands to prevent the rise of the water table, waterlogging, and salinity buildup in the soil. Although salt accumulation in receiving waters is the most prevalent problem affecting downstream users, the effect of irrigation and improved drainage on loss of trace elements to the environment has had the greatest impact in the U.S. These detrimental effects often can be avoided by identifying a reliable drainage outlet prior to construction of irrigation projects (Skaggs et al., 1994).

Applicability
CD structures require that the topography be relatively flat. The costs to production and water quality will usually exceed benefits when the land slope exceeds 0.5%.

Example 14

M379. Sediment removal from shallow lakes

Description of the problem
Need to restore aquatic ecosystems as well as the water storage capacity of the landscape.

Description of the measure
Restoration aims of sediment removal include the following objectives (Peterson, 1981):

1) Reduce internal phosphorus loading from relatively recently deposited nutrient rich sediment;
2) Increase water depth for water storage and navigation purposes;
3) Enhance fish production;
4) Reduce the abundance of emergent aquatic plants;
5) Maintain or create sufficient water depth for submerged water plants;
6) Remove toxic substances associated with the sediment.

Review of more than 60 projects and examination of 5 case histories (Lake Trummen, Sweden; Lake Herman, South Dakota; Wisconsin Spring Ponds; Steinmetz Lake, New York; and Lilly Lake, Wisconsin), reveals that the first four objectives are usually met through sediment removal.

Benefit in current climate
Contribution to good ecological status (GES) and avoid damage caused by floods was noticed by Belgium (Nixon, 2008).

Contribution to adaptation to climate change
The frequency of flood events is expected to increase as a result of climate change (Win-win, BE). Increased water storage capacity of the drainage basin helps to alleviate floods and provides water resources to overcome droughts.

Potential conflicts and problems under changing climate, precaution level

Dredging of lake sediments will remove rooted aquatic plants, however, their re-encroachment rate will depend on depth, sediment texture, and sediment nutrient content. Sediment removal to control toxic materials is possible with minimal environmental impact when proper equipment is used, but it may be extremely expensive.

Applicability

If considered possible by environmental impact assessment.

2.3 Keep substances at source avoiding them becoming pollutants

Landscape ecological processes can be sustainable only if the necessary physical and chemical provisions of the site are maintained. Soil losses of carbon and/or inorganic matter through leaching and erosion degrade terrestrial ecosystems and create heavy loadings to aquatic ecosystems where these compounds, especially nutrients, become pollutants. Climate change altering the temperature regime and precipitation patterns accelerates also the cycling of toxic substances (Eisenreich, 2005). There are several, often rather simple ways to diminish the losses and support the resilience of ecosystems by keeping substances in place in the landscape. Broadly the processes of water related matter losses from landscapes can be divided into leaching of dissolved compounds and erosion of solids by water.

2.3.1 Leaching

Ripl & Hildmann (2000) analysing published data showed high salt losses between 500 and 1500 kg ha^{-1} y^{-1} in most river catchments all over central Europe. The loss of mineral ions is irreversible unless they are precipitated and retained either in streams, lakes, or wetlands. The authors called the observed continuous depletion of the base cations from the most readily soluble salts in the catchments „landscape ageing“ and indicated the insufficient dissipation of solar energy in physical and biological cycles as the main driving force behind the accelerated leaching process. They showed that the evaporation-condensation processes which are the the main energy dissipating processes within a healthy landscape and support the short water cycle, have been disturbed by deforestation and thus more energy is not being turned over 'harmlessly' in these cycles.

Diffuse nutrient loading has become one of the most persistent obstacles for water quality management and restoration of water bodies (e.g., Baginska et al., 2003; Salvetti et al., 2006). The acceleration of the hydrological cycle due to climate change has resulted specifically in a more rapid transfer of nitrogen through river drainage basins (Huntington, 2006).

Bouraoui et al. (2009) analysing the factors responsible for nutrient losses into waters in 17 catchments in Europe covering a wide range of climatic conditions, soil and geological characteristics, found that the concentration of phosphorus was positively correlated to the rainfall intensity and the population density, while the nitrogen concentration was positively correlated to the agricultural surface. Both phosphorus and nitrogen concentrations decreased with the increasing proportion of surface waters in the catchment, indicating that lakes and reservoirs may contribute to the nutrient retention. In 44 watersheds in western Oregon, Floyd et al. (2009) observed strong negative correlations (r=–0.81 to –0.94) between nitrate-N in the runoff water during winter and spring and the proportion of woody vegetation in the influence zone of the surrounding stream networks. The marked acceleration of the global nitrogen cycle is the direct consequence of the increased application of chemical fertilizers over the last 60 years, which have doubled the reactive nitrogen in terrestrial ecosystems (Huntington, 2006) and may exhaust the denitrification capacity of aquatic ecosystems. Hefting et al. (2003) found that increasing nitrate load to riparian buffer zones decreased nitrogen buffering capacity but increased dramatically the emission rate of nitrous oxide.

2.3.2 Erosion

Water erosion as the displacement of solids by flowing water is a natural process, but it has been dramatically intensified by human land use, especially industrial agriculture, deforestation, and urban sprawl (Montgomery, 2008). Erosion exceeds topsoil regeneration rates by 16-300 times. About 0.7% of the world's topsoil is lost annually, that means 30% of topsoil will be lost by 2050 unless erosion is slowed or halted (Goodland, 1997).

A large proportion of nutrients lost from the catchment are bound to suspended solids and follow the erosion pathways. From arable lands the bulk of the erosion and phosphorus loading to surface waters originates outside the growing season while the condition in which the fields remain after harvest is crucial (Puustinen et al., 2007). Erosion and nutrient runoff from fields can be reduced by agricultural best management practices such as increasing vegetation coverage on arable land in winter (**M075**), mulching (**M091**), catch crops (**M066**) and late ploughing (**M070**). Water-saving measures in irrigation (**M186**) such as subsurface drip irrigation (Lamm & Camp, 2007) or using the repeated short sprinkler watering avoid the formation of surface runoff and erosion fluxes. The already mobilized suspended solids e.g., by torrential rain can be retained by creating buffer zones and buffer strips between agricultural land and surface water bodies (**M067; M083; M298**), grassing of arable land, in particular along watercourses (**M151**) and capturing polluted runoff from steadings in constructed farm wetlands (**M335**). The basic idea is to control the peak runoff and retain the water for a certain time to allow the suspended solids to settle or be filtered by the soil root system.

Also in forestry, the peak runoff control (PRC) method (**M376**) has been used with good results in temperate forests (Amatya et al., 2003), at peat harvesting sites (Marttila & Kløve, 2009) and in drained peatland areas (Marttila & Kløve, 2010). In peatland forestry the PRC method reduced suspended solids (SS) load by 86% by reducing flow velocities and improving settling conditions in the ditch network. The method had a considerable effect on SS-bound nutrients, reducing total nitrogen (Ntot) load by 65% and total phosphorus (Ptot) load by 67% (Marttila & Kløve, 2010).

Proper stormwater management in urban landscapes can considerably limit contaminant generation and transport. Due to fast runoff generation in urban areas, the first rainfall after a long dry period is the most polluted and need to be cleaned at the sewage treatment plant, while later, if rains continue, the water can be discharged directly to receiving water bodies. A system allowing separate treatment of the 'first flow' stormwater (**M377**) was developed in Reiderland under the LIFE Environment Programme 1992-2006 (LIFE, 2006). Given the highly dynamic nature of stormwater quality, McAlister et al. (2003) stressed the importance of using suitably small temporal resolution in stormwater quality modelling and continuous simulation over one or more years.

2.3.3 Urban and industrial wastes

Keeping substances in place is the main principle also in waste management. The biologically treated wastewater from villages and cities still contains a great amount of base cations and organic matter. Instead of burdening water courses, water should be returned to the landscape via polders. In such polders the production of biomass (reed, *Phalaris*, *Glyceria*, willows, etc.) can be used as a renewable resource (Ripl & Hildmann, 2000).

If flood frequencies increase, it will increase overflow frequencies of combined sewer systems and pose a threat to good surface water quality. It has been proposed to establish action plans for reducing overflow frequencies (**M104**) and avoid planning water treatment or sewage treatment infrastructures within floodplains (**M006**; Example 15).

In industry that uses a vast range of chemicals, part of which highly toxic, it is increasingly vital to control pollution load at source. Corresponding technologies are called end-of-pipe technologies, clean technologies, or closed cycle technologies. Proper management of industrial accident risk (**M001**; Example 16) should diminish environmental disasters.

Example 15

M006 Avoid planning water treatment or sewage treatment infrastructures within floodplains

Description of the problem

The nature of water treatment plants and sewage treatment plants is that they are located close to major rivers in order to abstract water from them or to discharge treated sewage effluent to them. It is therefore to be expected that the plants will have a certain level of flood risk. A significant flood at a water treatment plant could result in the contamination of drinking water supplies by flood water, the risk of this would lead to the shutting down of the plant. A significant flood at a sewage treatment plant could result in the contamination of rivers and land as the flood spreads untreated or partially treated sewage and effluent from the works. The operation of plants may also be affected by ancillary power losses. In addition to the above listed water works there may also be pumping stations and other installations that relate to water supplies and distribution infrastructure[19].

Description of the measure

The UK policy statement on flood risk[20] allocated water treatment works within the "less vulnerable" land use classification. This means that new water works development should not be located within the functional flood plain (flood zone 3b)

Benefit in current climate

No clear benefits.

Contribution to adaptation to climate change

Decreases in case of flooding the contamination risk of drinking water supplies by flood water and the contamination of rivers and land by untreated or partially treated sewage and effluent from the works.

Potential conflicts and problems under changing climate, precaution level

The measure may present difficulties as there will be a need to propose new development at existing works in order to improve the capacity or quality of the water treatment. UK government is currently consulting on a proposal to transfer water treatment works into the Essential Infrastructure classification meaning that it would be possible to locate new development within Flood Zone 3b providing the Exceptions Test was passed.

Applicability

In most flood plain areas with a certain flood risk level.

Example 16

M001 Managing industrial accident risk

Description of the problem

Industries present a potential risk of surface and groundwater contamination with toxic substances such as heavy metals, persistent organic substances, and inadequately treated industrial discharges. These can have a toxic effect on animals and plants and/ or accumulate within the food chain and in sediments

[19] London Regional Flood Risk Appraisal, October 2009 http://www.london.gov.uk/mayor/strategies/sds/docs/regional-flood-risk09.rtf

[20] PPS25. Planning Policy Statement 25: Development and Flood Risk, December 2006, London, TSO, available at: http://www.communities.gov.uk/publications/planningandbuilding/pps25floodrisk

(SEPA, 2007). Contamination risk increases during geohazards like floods, earthquakes, tornados, and tsunamis.

Description of the measure

Prevention of significant losses of pollutants from technical installations and/or reduction of the impact of accidental pollution incidents requires a **complex of measures** specific for the type of industry and the geohazard risk level of the location.

Benefit in current climate

Reduction of toxic contamination risks.

Contribution to adaptation to climate change

Improves preparedness to combat flood impacts.

Potential conflicts and problems under changing climate, precaution level

No

Applicability

Unlimited. Finland has included this as a win-win measure into the program of measures (PoM) of the 1st RBMP.

2.4 Keep species within their natural habitats

Discussing the aspects of rapidly decreasing biodiversity, one of the founders of the ecological sustainability concept, Robert Goodman (1995) wrote: "*Although biodiversity conservation is becoming a general ideal for nations and development agencies, there is no agreement on how much should be conserved, nor at what cost. Leaving aside the important fact that we have not yet learned to distinguish useful from non-useful species, agreeing on how many other species to conserve is not central to the definition of environmental sustainability. Reserving habitat for other species to divide among themselves is important; let evolution select the mix of species, not us."*

The principle that species can effectively be protected only through protecting their habitats has been the basis for the EC Habitat Directive (EC, 1992). By the time this directive was adopted, Europe's natural habitats continued to deteriorate and an increasing number of wild species were seriously threatened mostly as a result of and agricultural intensification and urban development. These factors remain the leading factors threatening biodiversity, however, climate change starts adding to the pressures.

Hickling et al. (2005; 2006) showed that a wide variety of vertebrate and invertebrate species have moved northwards and uphill in Britain over approximately 25 years. Many species are potentially endangered (Hering et al.,

2009) by climate change or, in contrary, are expected to expand their distribution areas in Europe (Ficetola et al., 2009).

Habitat and biodiversity protection in the context of climate change should include *inter alia* eventual restoration of habitats lost through sea level rise (**M305**) or increased flooding (**M301**), monitoring and adjusting abstractions and other pressures which reduce river flows (**M278; M308**; **M313**) and groundwater levels (**M304**) for groundwater dependent and/or supported habitats and species, reducing of habitat fragmentation (**M307; M329**), protection and restoration of wetlands (**M032**, Example 4; **M295**), rivers (**M053; M061**), and floodplains (**M294**).

Franklin et al. (2002) defined habitat fragmentation as the discontinuity in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, or survival in a particular species. Pressures to populations caused by habitat fragmentation can be alleviated by measures such as fish ladders (**M063; M355**), bypasses, and culverts (**M002**, Example 17). The general principle should be to keep the natural but remove manmade barriers. The dyke system of the North Sea coast in The Netherlands is a special case where the increase of saltwater-freshwater connections for the benefit of fish migration (**M047**) has been considered a regret measure due to the loss of freshwater volumes which might be valuable for agriculture, wildlife and as drinking water in times of low discharges and low precipitation (Nixon, 2008).

Keeping the natural barriers applies for measures dealing with invasive species, controls on their importation and introduction (**M101; M206**; **M327**).

Example 17

M002 Bypasses and culverts (roads, railway) to improve fish migration

Description of the problem

Complex life-cycle of diadromus fishes, such as several salmonids, involves the migration of juvenile fish from freshwater to the sea and the migration of adults from the sea to freshwater spawning grounds. Other major migratory fish species include eel, lamprey and shad. In order to sustain fish populations, the importance of allowing free movement also of coarse fish species has increasingly been recognized. Many man-made obstructions such as dams, weirs and mills, restrict this access to spawning areas.

Description of the measure

A fish pass (Fig. 4), also known as a fishway, fish ladder or fish steps, is a structure on or around artificial barriers to facilitate fishes' natural migration. Most fishways enable fish to pass around the barriers by swimming and leaping up a series of relatively low steps (hence the term ladder) into the waters on the other side.

Fig. 3 *Stone culvert*
Source:
http://canal.mcmullans.org/culvert.htm

Fig. 4 *Fish ladder*
Source: www.linkelconstruction.com

A culvert (Fig. 3) is a conduit used to enclose a flowing body of water. It may be used to allow water to pass underneath a road, railway, or embankment, for example, and to enable fish migration. Culverts can be made of many different materials, such as steel, plastic, concrete or stones.

Benefit in current climate

Bypasses and culverts, decrease the fragmentation of aquatic ecosystems and contribute to their normal functioning that is a basic aspect of ecological water quality.
In forestry, proper use of cross-drainage culverts can improve water quality while allowing forest operations to continue[21].

Contribution to adaptation to climate change

As far as their flow capacity is not exceeded, culverts represent a most wide spread type of flood protection structures.

Potential conflicts and problems under changing climate, precaution level

Culverts should be dimensioned large enough in order to not increase the natural flood stage in case of potential flood hazards. The recommended practice is for the designer to select appropriate hydrologic estimating procedures, and obtain runoff data where available for purposes of evaluation, calibration and determination of the predicted values for the desired flood frequencies (ConnDOT, 2001).

Applicability

General. Due to restricted flow capacity with increased water flow and extreme floods, Norway has indicated construction of culverts as a potential counterproductive measure (Nixon, 2008).

[21] http://www.extension.umn.edu/distribution/naturalresources/DD6979.html

3 Keep things natural

Keeping things natural means protecting and restoring the natural regulating function of catchments, rivers, floodplains and coasts in order to manage water quality and to alleviate flood and coastal erosion risk. This could involve diverse actions such as flow modification (**M051; M352**), floodplain reconnection (**M019** – Example 5; **M024; M025**– Example 9, dam removal (**M004** – Example 18) instream and coastal habitat improvement (**M301**), and riparian management (M065; M150). Restoring degraded peat bogs (**M378**) and reforestation (**M029** – Example 2) will also help to slow run-off and increase infiltration. Sustainable urban drainage systems (**M005** – Example 11) follow the same spirit of naturality in urban areas.

Hydraulic modifications of rivers to reduce flood damages have a long history. Construction works were undertaken to prevent overbank waters and ensure unrestricted flow of flood volumes. For these purposes, rivers were straightened, channelized, and squeezed between embankments, disregarding the natural dynamics of the river and its ecosystem. According to a cartographic study in mid 1980s (Brookes, 1984), only about 900 km out of the 30,000 km of Danish watercourse of natural origin had retained their natural form. The main shortcomings of such river modification approach were summarized by Poulard et al. (2010) in the following three issues:

1. acceleration the flow often results in aggravating floods downstream,
2. the disruption of the natural patterns can disrupt the sediment balance, hence causing erosion or deposits,
3. the consequences on ecosystems are often disastrous.

Geilen et al. (2004) and Poulard et al. (2010) have analysed effective flood-protection solutions with alleviated impacts on the ecosystems and advocate a close cooperation between biologists and hydraulic practitioners for finding best measures. The best practice document on flood prevention, protection and mitigation (EU, 2004) includes as the first basic principle: *As far as possible, human interference into the processes of nature should be reversed, compensated and, in the future prevented. It is necessary to promote and harmonise changes in water policies and land-use practices, as well as environmental protection and nature conservation, in order to improve flood management in the frame of Integrated River Basin Management.*

Since 1991 when the first international conference on river restoration was held in Lund, Sweden (Osborne et al., 1993), there has been increasing interest in Europe in rehabilitation of watercourses and river valley ecosystems. Still there are rather few examples in Europe so far of restoring functional floodplains. A recent overview of river restoration projects in Europe (Moss & Monstadt, 2008) explores the reasons behind this discrepancy between interest and applications explaining it with institutional constraints.

Early floodplain restoration schemes were undertaken in the mid-1990s in the Rheinvorland-Süd on the Upper Rhine (Bissels et al., 2004), the Bourret on the Garonne (Aguilar-Ibarra et al., 2005), and the Long Eau River project in England (Moss & Monstadt, 2008). Some of the ongoing projects include Lenzen on the Elbe (Neuschulz & Purps, 2003), La Basse on the Seine (Ducrotoy & Dauvin, 2008) and the Parrett Catchment Project in England (Somper, 2005). The spatial planning project "Room for the River" in The Netherlands (Spatial, 2006), which included a number of measures leading to improvement of stream morphology and floodplain restoration was initiated in 2006 and will be ongoing until 2015.

In 1994, the project "Watercourse restoration – Methods and effects" was initiated aiming to collect and collate existing knowledge in Denmark on restoration methods and their effects which were published as a book two years later (Hansen, 1996). To benefit most from the knowledge and experience collected in various European projects on river restoration, the **European Centre for River Restoration** (ECRR), a non profit organization, was established in 1999 in Silkeborg (Denmark) within a LIFE Environment programme that ensured funding during the 1999-2002 period. For the first three years the secretariat was held by the National Environmental Research Institute (NERI, Denmark), for the following four years, by the Institute for Inland Water Management and Waste Water Treatment (RWS-RIZA, The Netherlands), from mid 2006 until the end of 2009 the Italian Centre for River Restoration (CIRF, Italy), and since January 2010, the ECRR secretariat is hosted by DLG Government Service for Land and Water Management in The Netherlands. A recent study under the EU FP6 Project Euro-limpacs (Jähnig et al., 2010) compared 26 pairs of restored and unrestored river sections in Austria, Czech Republic, Germany, Italy and the Netherlands evaluating the restoration success. The study found significant improvement in the diversity of mesohabitats in 83% of the studied river sections. For microhabitats, restoration had a significant positive effect on diversity at 69% of sites but a significant negative effect at 15% of sites.

In the United States, **National River Restoration Science Synthesis** (NRRSS) Project has collected information on 37,099 projects in the USA (Bernhardt et al., 2005). The aim of the synthesis was to:

1. Evaluate the state of the practice of stream restoration nationally and identify successful demonstrations of different types of stream restoration, highlighting the reasons for their success.
2. Produce a scientific document that examines the links between ecological theory and stream restoration (such as the roles of refugia, connectivity, and natural processes), and identifies the unanswered questions meriting further research.
3. Develop a series of specific recommendations to improve how stream restoration is carried out and its success evaluated.
4. Disseminate this information broadly and on an on-going basis.

The restoration goals were split into the following categories:

- Aesthetics/recreation/education (e.g., trash removal),
- Bank stabilization (revegetation, bank grading),
- Channel reconfiguration (bank or channel reshaping),
- Dam removal,

- Fish passage (fish ladders),
- Floodplain reconnection (bank or channel reshaping),
- Flow modification (flow regime enhancement),
- Instream habitat improvement (adding boulders/woody debris),
- Instream species management (native species reintroduction),
- Land acquisition,
- Riparian management (livestock exclusion),
- Stormwater management (wetland construction).

Example 18

M004 Removal of (hydropower) storage dams

Description of the problem[22]

Many medium size dams in Europe that have been built from the beginning of the century to the end of World War II are now reaching the end of their lifetime. Most of these dams are located in mountainous areas and especially in the Alps (Switzerland, Italy, France, Austria) and in Norway. They are 3 to 25 meters high or more and were built for electricity or, less often, for water – supply.

After World War II, dam projects were more and more important and were located both in mountainous areas and on lower parts of rivers (and even sometimes on estuaries).

In most European countries (with the exception of some Eastern countries, and the ex-Soviet Union) almost every dam is under a concession which lasts from 40 to 60 years. This period is usually smaller than the physical lifetime of the building.

Description of the measure[23]

There exists a range of options for renaturalising dams — from the staged removal of dam structures, to partial modification. There is no generic approach to river restoration. A solution is needed that addresses the particular characteristics of each river and dam.

Dam dismantling

This is the most dramatic option, involving the complete dismantling of all physical barriers to stream flow. The intention here is to fully restore the natural flow of the river, including peak flows and seasonal flooding. This would also enable fish passage and the transport of gravel and organic debris downstream. Dam removal can sometimes be immediate, but more often it is staged in a cautious, risk-averse way to avoid unwanted release of the sediments that typically accumulate behind old dams.

Dam Decommissioning

This option alters the dam structure, restores flow, and permanently changes the dam's original function. However, some of the dam may be left intact, recognizing that complete removal of dams may not always be the best option for a river. For example, remnant structures may serve to stabilize reservoir

[22] http://www.rivernet.org/general/dams/decommissioning/decom3_e.htm#context
[23] http://commons.bcit.ca/recovery/decom.html updated 07/29/2008

sediment, or provide a limited buffer against flooding. Also, partial alteration helps to avoid the expense of complete removal. Dam decommissioning can provide these benefits while still achieving the ecological objectives of improved fish passage and greater instream flows.

Dam Modification

There are other options that have little or no impact on dam function, allowing existing dams to continue providing societal benefits such as electricity and drinking water. For example, the addition of fish ladders usually improves fish access to spawning habitat above the dam without altering the function of the dam itself.

Benefit in current climate

Reduction of hydromorphological pressure has been indicated by Slovenia (Nixon, 2008).

Contribution to adaptation to climate change

Dam modification and more effective management of dams can help to mitigate environmental impacts, e.g. maintain river flow and improve fish survival downstream by releasing more water from the dam reservoir during critical times such as spawning season and drought.

Potential conflicts and problems under changing climate, precaution level

Removal of storage dams may increase flood risk.

Applicability

In western and northern Europe an estimated minimum of ten thousands or more dams higher then approximately 3 m require a renewal of their concession during the next 10 - 20 years. The figures for eastern and southern Europe are unknown yet[22].

There will therefore be, from 2010 on, another important number of big dams built from the 1950s to the late 1980s whose concession will draw to a close[22].

Since most of the concessions are drawing to a close only now, European experience in handling this problem is rather limited. First lessons have been learned by France[24] and Spain[25].

Dam removal is not always realistic or feasible. An alternative to dam removal is to simulate periodic flood pulses consistent with historical magnitude and timing by releasing large amounts of water at once instead of maintaining more consistent flows throughout the year. This would allow overbank flooding, which is vital for maintaining the health of many riparian ecosystems (Bhattacharjee et al., 2009).

---

[24] http://www.rivernet.org/general/dams/decommissioning/pdfetdocs/edf_damdemoval_experiences.pdf
[25] Liberando ríos. Propuestas de WWF para el desmantelamiento de presas en España. http://assets.wwf.es/downloads/presas_informe_completo.pdf

4 Be informed and plan your actions

A large and heterogeneous group of measures deals with administrative issues, planning, and capacity building in the sense of research, education and stakeholder envolvment. According to temporal scale, these issues can be divided into long-term (most of strategic planning, research and education measures), medium-term (adaptive planning in the RBMP cycles) and short-term or operative issues, such as flood alert systems.

In a global change context, several recent advances in the field of hydrology and biogeochemistry suggest that a move from a riparian to a river drainage basin perspective is necessary to reframe research and thus provide a more integrated scientific understanding to inform water- and land-use management and policy (Pinay & Hannah, 2009). For example, the significant effect of land cover on river flows becomes evident only with increasing spatial (basin area) and temporal (seasonal, annual, and beyond) scale (Bloshl et al., 2007). Watersheds have been considered useful and globally applicable management units in which context to analyze and debate issues related to social and inter-generational health and equity, environmental change and social–ecological resilience (Parkes et al., 2010). Proposed measures like 'Implementation of river basin management plans' (**M013** – Example 19), 'Integrated coastal zone management' (ICZM; **M130**) or 'Development of management plans water resources in drought conditions' (**M213**) may sound too complex to be called measures, but this characterises the real situation with adaptation measures in Europe.

Community based adaptive management is the preferable way of watershed governance as it integrates social and ecological suitability to achieve conservation outcomes by providing landowners the flexibility to use a diverse set of conservation practices to achieve desired ecological outcomes, instead of imposing regulations or specific practices (Habron, 2003).

4.1 Uncertainty and the precautionary principle

Biophysical linkages in complex self-regulating systems are inherently uncertain that makes important considering the precautionary principle while making management decisions in these conditions. Because of the uncertain character of practically all the global life-support systems, Goodland (1995) calls to be very conservative in our estimate of various input and output capacities, and particularly of the role of unstudied, apparently "useless" species, because "*…better to be roughly right than precisely wrong /…/ In cases of uncertainty, sustainability mandates that we err on the side of prudence*". Adaptive management approach thrives on information collection and use, but it also enables action in the face of information shortage identifies uncertainties and establishes methodologies to test hypotheses concerning those uncertainties.

4.2 Long term capacity development

4.2.1 Research needs

Numerous authors have called for research to reduce uncertainty over (a) how climate change may affect freshwaters and (b) how water- and land-use managers should mitigate and adapt to climate change (Prudhomme & Davies, 2009). Although the future-oriented nature of any planning process remains uncertain, research can to some extent decrease the uncertainty by filling knowledge gaps. All climate change and adaptation strategies contain as a basic principle the research needs: to improve the temporal and spatial resolution of climate projections and to advance our knowledge on the relationships between climatic variations and water resources (**M287; M322**), ecosystems (**M207**), flood risk (**M273**), and pollution spreading (**M113**). Pinay & Hannah (2009) showed that uncertainties in predicting impacts may be attributed to limitation of historical data (in terms of duration, spatial coverage, homogeneity and so on) for model parameterisation, calibration, and validation; incomplete knowledge of complex process nonlinearity and feedbacks; general circulation model (GCM) scenarios; downscaling of GCM data to basin scale; and hydrological models. There is a need to develop methodologies for assessing potential damage of flood risk areas (**M219**). Better climate change projections are especially important for planning large infrastructures like dams (**M195**).

The research results can be generalized as maps showing the current and future climate spaces and the vulnerability and impacts for priority species and environments (**M330**), zones of flood (**M270**) or landslide hazard (M119). These maps represent powerful tools in water management at different time scales and organisational levels. For example, classification of river basins in water scarcity or water deficit (**M012** – Examples 20 & 21), implies certain management and social security schemes to be applied.

A recent paper by Wilby et al. (2010) examined the scientific basis for adaptively managing vulnerable habitats and species. Expert and policy-maker responses were grouped into six adaptation supporting activities:

1. detecting climate change impacts;
2. managing multiple anthropogenic pressures;
3. restoring riparian vegetation;
4. assessing and protecting environmental flows;
5. managing transitions to new ecosystem states;
6. integrating and appraising adaptation options.

Although the title was „Turning adaptation principles into practice", the review did not go further from the knowledge base issues for adaptation. The measures were numbered as **M409-M451** in the present database.

4.2.2 Education

Adaptive management uses management as a tool not only to change the system, but as a tool to learn about the system. It is concerned with the need to learn and the cost of ignorance, while traditional management is focused on the need to preserve and the cost of knowledge[26].

If in Europe the need for education and advice to ensure efficient adaptation is stressed mostly at farm and regional scales (**M015** – Example 22; **M016**), education in issues related to water saving and protection from pollution becomes vital in arid and drought prone areas. Environmental educational programmes (**M158**) such as the Worldwide Water Education Project WET[27], which publishes water resource materials in several languages, provides training workshops on diverse water topics (i.e., watersheds, water quality, water conservation), and organizes community water events for children, parents, teachers and community members, turns water education into a water management tool.

4.3 Medium-term management

Measures within the time frame of a river basin management cycle of 6 years can be based on rather solid climate projections, although unpredictable extreme situations may divert their efficiency. Anyway, these measures deal with rather concrete targets and were numerously represented already in the 1st RBMP round. These measures were aiming at certain water resources regulation schemes (**M045; M050; M163, M268**), prioritization (**M008** – Example 23; **M041; M088; M208**), water saving (**M242; M085; M186**), metering (**M007** – Example **M096; M306**), abstraction and discharge licencing (**M014** – Example 24; **M110**) and pricing (**M044; M242**). According to the guidance on river basin management in a changing climate (CIS, 2009), in general, reference conditions and default objectives should not be changed due to climate change projections over the timescales of initial WFD implementation (up to 2027) unless monitoring reveals long term coherent changes in the status of reference water bodies over large geographical areas. This eventual adjustment of reference conditions and setting the quality objectives in some water body types is also possible within the RBM planning cycle (**M193**).

4.4 Operative measures

Difficulties with flash flood observations, inefficient hydrometeorological data transfer and lack of an archive of flood events hinder the development of a coherent framework to analyze flood hazard and vulnerability at the pan-European scale (Barredo, 2007). According to the estimate of Handmer (2001), between five and ten percent of Western Europe's population lives or works in floodplains and even more people are exposed to flood risks because recreation and transportation facilities are also flood prone. This makes the need to develop and apply efficient flood-warning systems a need. A number of measures aim at development and modernization of information systems of the flood forecasting and warning service (**M173; M205; M221**) and early warning systems in areas

[26] http://www.resalliance.org/index.php/adaptive_management
[27] http://projectwet.org/

with low slope stability (**M119**). It is also important to train the use of the early warning systems (**M222**).

4.5 Streamlining of strategies and avoiding potential cross-sectoral trade-offs in river basin management

Climate change affects nutrient and carbon losses from terrestrial ecosystems and their loads into aquatic ecosystems. For mitigating nutrient losses/loads, river basin management should plan better matching of nutrient supply with plant demand. Climate change mitigation measures aim to reduce greenhouse gas (GHG) emissions while adaptation measures should reduce the vulnerability of societies and ecosystems to adverse effects of climate change. In respect of water resources and ecological status of water bodies the two approaches are often disconnected that, instead of synergies, can create trade-offs between them. It is well-known that large-scale biofuel production increases water demand and contamination, hydro-electric power plants fragmentise the river ecosystem integrity and affect biodiversity (see Example 1), dams and water reservoirs can emit additional GHGs, and seawater desalination as a drought combating measure accelerates energy consumption. It is much less known that even reforestation (Example 2), wetland reconstruction (Examples 3 & 4), floodplain restoration (Example 5) or creating buffer strips, usually considered as win-win measures (Nixon, 2008), may locally become antagonistic to other adaptation and mitigation measures (e.g., Jackson et al., 2005). Careful spatial planning should avoid trade-offs between mitigation and adaptation, and make it possible to combine the reduction of vulnerability with mitigation of GHG emissions. Environmental impact assessment and strategic environmental assessment should be applied to analyze the environmental effects of proposed measures and to find an optimal prioritisation of the Multiple Uses and Functions of Water Services (MUFS).

Example 19

M013 Implementation of river basin plans

Description of the problem

The close inter-linkages between the hydrological, ecological and socio-economic components of river basins have rarely been given adequate consideration by decision-makers such as politicians, land-use planners and water engineers. There is enough freshwater in the world to meet present needs and accommodate growing populations. The challenge is to protect the sources of freshwater and manage its use in a manner that is both equitable and ecologically sustainable.

Description of the measure

Integrated river basin management (IRBM) is the process of coordinating conservation, management and development of water, land and related resources across sectors within a given river basin, in order to maximise the economic and social benefits derived from water resources in an equitable manner while preserving and, where necessary, restoring freshwater ecosystems (GWP, 2000).

IRBM is a **holistic approach** that rests on the principle that naturally functioning river basin ecosystems, including accompanying wetland and groundwater systems, are the source of freshwater. Therefore, management of river basins must include maintaining ecosystem functioning as a paramount goal. This **'ecosystem approach'** is a central tenet of the Convention on Biological Diversity.

River basins are dynamic over space and time, and any single management intervention has implications for the system as a whole.

The **seven key elements** to a successful IRBM initiative are (GWP, 2000):

1. A long-term vision for the river basin, agreed to by all the major stakeholders.
2. Integration of policies, decisions and costs across sectoral interests such as industry, agriculture, urban development, navigation, fisheries management and conservation, including poverty reduction strategies.
3. Strategic decision-making at the river basin scale, which guides actions at sub-basin or local levels.
4. Effective timing, taking advantage of opportunities as they arise while working within a strategic framework.
5. Active participation by all relevant stakeholders in well-informed and transparent planning and decision-making.
6. Adequate investment by governments, the private sector, and civil society organisations in capacity for river basin planning and participation processes.
7. A solid foundation of knowledge of the river basin and the natural and socio-economic forces that influence it.

Practical solutions for managing rivers better include (GWP, 2000):
- Protected areas to safeguard sites such as headwaters and wetlands that contribute to maintaining water quality and quantity.
- Forestry practices that are compatible with protection of freshwater resources.
- Sustainable agriculture that takes advantage of local conditions, uses less water and is not so dependent on chemical pesticides and fertilisers.
- Improved performance of water intensive industries.
- Innovations in the design of shipping so that fewer alterations to natural river channels are required for commercial navigation.
- Dam and reservoir operations that mimic natural flow regimes.
- New technologies that reduce water consumption by sanitation and energy production processes.
- Restoration techniques to re-establish valuable natural functions in heavily degraded freshwater systems.

Yet none of these tools will be effective in isolation. Indeed, if one solution is pursued while other issues or sectors are ignored, the effects are at best strictly localised and at worst temporary and ultimately futile.

There have been some critical remarks to the river basin approach from social sciences arguing that "...geography and hydrology do not necessarily define the best scale for planning and problem solving" (Rhoades, 1998) and that "…watersheds as closed human management units are external, bureaucratic or researchers fantasies not indigenous ones" (Winpenny, 1994).

Benefit in current climate
Integrated river basin management (IRBM) provides the framework in which the full range of tools and approaches can come into play, with multiple sectors working together, rather than at cross-purposes, to manage and conserve freshwater resources sustainably and equitably.

Contribution to adaptation to climate change
RBMPs develop a long-term vision for the river basin, agreed to by all the major stakeholders. Climate change has the potential to undermine decisions and investments made within river basins. This means that it needs to be ensured that measures are flexible and robust enough to be viable under changing climate conditions and that they do not run counter to adaptation or mitigation objectives. Although the EU Water Framework Directive (WFD, 2000), the leading document in river basin management in Europe, does not explicitly mention risks posed by climate change to the achievement of environmental objectives, there is a strong case for incorporating climate change within the RBM planning process. In particular, the integrated approaches to land, water and ecosystem management, combined with the cyclical review of progress, are all consistent with the ideals of **adaptive management**.

Potential conflicts and problems under changing climate, precaution level
Following the IRBM principles should avoid conflicts.

Applicability
Universal.

Example 20

M012/1 Classification of river basins in water scarcity

Description of the problem (IWMI, 2006)
Water stress and **water scarcity** occur when the demand for water exceeds the available amount during a certain period or when poor quality restricts its use. According to the **Falkenmark Water Stress Indicator** (Falkenmark & Lindh, 1976), a country or region is said to experience "water stress" when annual water supplies drop below 1,700 m^3 $pers.^{-1}$ y^{-1}, at levels between 1,700 and 1,000 m^3 $pers.^{-1}$ y^{-1}, periodic or limited water shortages can be expected. When water supplies drop below 1,000 m^3 $pers.^{-1}$ y^{-1}, the country faces "water scarcity." This definition of scarcity—relating water availability to water demand—implies that dry areas are not necessarily water-scarce.

Scarcity can be **physical** (absolute), such as in environments of low precipitation and large evaportranspiration rates or **economic**, induced by economic or political constraints, which do not permit the adequate development of water resources. A quarter of the world's people live in areas characterized by physical water scarcity. One billion live in basins that face economic scarcity, where human capacity or financial resources are likely to be insufficient to develop adequate water resources (Fig. 5). An emerging alarming trend is an **artificially created scarcity** due to overdevelopment of hydraulic infrastructure, most often for irrigation. Water resources are overcommitted to various users, and there simply is not enough water to meet human demands and meet environmental flow needs.

Symptoms of physical water scarcity include severe environmental degradation including river desiccation and pollution, declining groundwater, problems of water allocation where some groups win at the expense of others. Around 900 million people live in river basins where the physical scarcity of water is absolute (the basins have closed). And another 700 million live where the limit to water resources is fast approaching (closing basins).

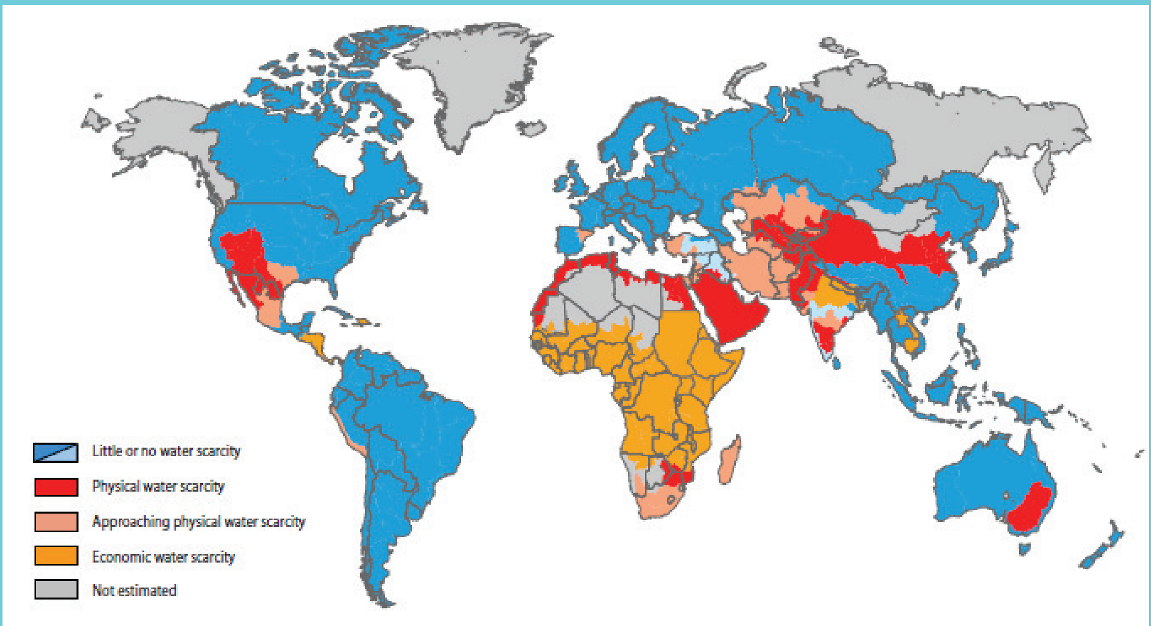


Fig. 5 ***Areas of physical and economic water scarcity***

Red: Physical Water Scarcity. More than 75% of the river flows are allocated to agriculture, industries or domestic purposes (accounting for recycling of return flows).

Light Red: More than 60% of river flows are allocated. These basins will experience physical water scarcity in the near future.

Orange: Economic Water Scarcity. Water resources are abundant relative to water use, with less than 25% of water from rivers withdrawn for human purposes, but malnutrition exists. These areas could benefit by development of additional blue and green water, but human and financial capacity are limiting.

Blue: Abundant water resources relative to use: less than 25% of water from rivers is withdrawn for human purposes.

Source: IWMI (2006)

Description of the measure
A world map on water scarcity was compiled by International Water Management Institute (IWMI) in 2006.

Benefit in current climate
Impose the implementation of a strategy to return to equilibrium between needs and resources.

Contribution to adaptation to climate change
Elaborated program of measures helps to avoid future crisis situations with water supply and adverse effects on agriculture and environment.
Potential conflicts and problems under changing climate, precaution level

Applicability
Global.

Example 21

M012/2 Classification of river basins in water deficit

Description of the problem
Soil water deficit is the amount of available water removed from the soil within the crop's active rooting depth. Likewise it is the amount of water required to refill the root zone to bring the current soil moisture conditions to field capacity. Soil water decreases as the crop uses water (evapotranspiration) and increases as precipitation (rainfall or irrigation) is added. Expressed in soil water deficit, evapotranspiration increases the deficit and precipitation decreases it. It is usually expressed in mm or inches of water and can be estimated by several methods[28].
The **water deficit index** (WDI) of Moran (Moran et al., 1994) is defined as

$$WDI = 1- E_a /E_0$$

where E_0 and E_a are the potential and actual evapotranspiration.
WDI varies from 0 to 1. WDI = 0 means that the land surface is extremely humid and covered by well-watered forest or water-saturated soil, and WDI = 1 means that the surface is in an extremely arid condition or completely covered by desert.

Two water balance parameters - actual evapotranspiration (AET) and deficit (D) - are biologically meaningful, are well correlated with the distribution of vegetation types (Fig. 6) and exhibit these qualities over several orders of magnitude of spatial scale (continental to local). Several well-known climatic parameters are biologically less meaningful or less important than AET and D, and consequently are poorer correlates of the distribution of vegetation types. Of particular

[28] http://www.extension.umn.edu/distribution/cropsystems/components/DC3875c.html

interest, AET is a much better correlate of the distributions of coniferous and deciduous forests than minimum temperature. The effects of evaporative demand and water availability on a site's water balance are intrinsically different. For example, the 'dry' experienced by plants on sunward slopes (high evaporative demand) is not comparable to the 'dry' experienced by plants on soils with low water-holding capacities (low water availability), and these differences are reflected in vegetation patterns. Many traditional topographic moisture scalars - those that additively combine measures related to evaporative demand and water availability are not necessarily meaningful for describing site conditions as sensed by plants; the same holds for measured soil moisture. However, using AET and D in place of moisture scalars and measured soil moisture can solve these problems (Stephenson, 1998).

Description of the measure

There are several methods available to measure soil water deficit[29]:

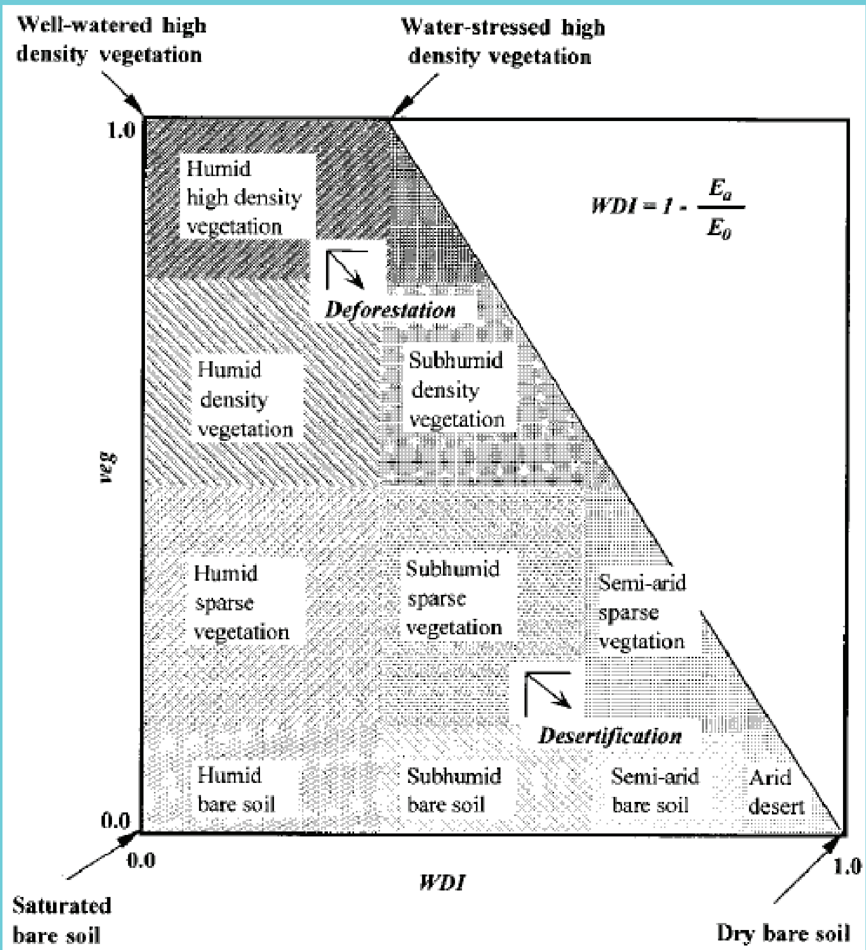


Fig. 6 *A theoretical scheme showing different land use types vs WDI and vegetation fraction (veg), and the direction of desertification and deforestation vs these two indices (Wang & Takahashi, 1999).*

- Soil Feel and Appearance

[29] http://www.extension.umn.edu/distribution/cropsystems/components/DC3875c.html

- Soil Water Sensors
  - Tensiometer
  - Electrical resistance sensors
- Soil Water Accounting

Benefit in current climate
Contributes to right crop selection and sustainable agriculture.

Contribution to adaptation to climate change
Active intervention in the composition of the vegetation is the main method for combating desertification (Tian et al., 2009).

Potential conflicts and problems under changing climate, precaution level
-
Applicability
Desertification risk areas, areas of irrigation agriculture.

Example 22

M015 Education and advice at farm and regional scales to ensure efficient adaptation

Description of the problem
The following **farm level aspects** need to be considered in climate change adaptation (Gibbons & Ramsden, 2008):
1. Farms are the level at which management decisions are made.
2. Farmers make decisions using information available to them.
3. Adaptation is not always instant, it may be gradual, there are degrees of adaptation and previous adaptations will influence what the farmer does currently.
4. Some farm-level investment decisions are costly and provide a flow of resources over multiple years (e.g. building a reservoir).
5. Different farmers will make different decisions using the same available data.

Farmers' choices are influenced by a number of factors, such as (Deressa et al., 2009):
- level of education,
- gender,
- age,
- wealth of the head of household,
- access to extension and credit,
- information on climate,
- agroecological settings.

The main barriers, however, include **lack of information on risks and adaptation methods** and financial constraints.

Description of the measure

While **general education level** of farmers helps them in accessing and understanding information, **specific information** should be given to farmers on (DFID, 2004):

1) **Weather and climate change forecasts/projections**
- Short-term weather forecasts, including cyclones and flooding prediction, will help farmers to decide when to plant or harvest crops but may also save lives and property.
- Medium-term, seasonal weather predictions can assist farmers to decide which crops or cultivars to grow.
- Long-term climate change scenarios of changing rainfall patterns, temperature and sea level rise can inform more strategic decisions about the approach and location of development, to plan migration, livelihood diversification or alternative land-uses.
-

2) **Short and long term vulnerability and hazard** related to climate change, including
- numbers of people affected,
- estimated economic impacts,
- planned financial aid mechanisms.

Climate information on its own does not always allow individuals, communities or governments to assess their risks or undertake risk reduction activities. Climate information needs to be combined with a number of other types of information to give an indication of risk.

3) **Common adaptation methods in agriculture** (Deressa et al., 2009):
- new crop varieties and livestock species better suited to drier conditions,
- irrigation systems, their efficiency and problems,
- possibilities of crop diversification,
- mixed crop and livestock farming systems,
- appropriate planting dates etc.

Benefit in current climate

Better education and information will contribute to better decision making.

Contribution to adaptation to climate change

Farmers with higher levels of education are more likely to adapt better to climate change because of their better access to information on improved technologies.

Potential conflicts and problems under changing climate, precaution level

Information must be up to date and corresponding to local conditions. Farming and adaptation practices that have proved useful in other areas should pass a careful test before being recommended or applied.

Applicability

Universal.

Example 23

M008 Solve quantitative unbalance

Description of the problem

Balancing the accessibility of water resources between various needs and deciding about prioritization schemes is a complex question that sharpens in areas of water stress, i.e. where the demand for water exceeds the available amount during a certain period or when poor quality restricts its use.

The 5th World Water Forum (Istanbul, March, 2009)[30] concluded that

- Instability of water delivery for irrigation creates a big risk for achievement of Millennium Development Goals (MDGs) not only for food, but also poverty alleviation because more than 50% of population in rural area is connected with irrigation;
- Hydropower became a principal competitor of food production and taking into account growth of prices on energy incomparable with prices of food, irrigation production is failing.

Description of the measure

Equitable allocation of water resources is one of the basic principles of Integrated Water Resources Management (IWRM). This implies improved decision-making, which is technically and scientifically informed, and can facilitate the resolution of conflicts over contentious issues. The catchment or watershed approach implies that water should be managed alongside the management of codependent natural resources, namely soil, forests, air and biota. There are existing tools (e.g. multi-criteria analysis) to help decision-making in terms of balancing social, ecological and economic considerations. These should be tested and applied (IWA, 2008).

The recognition of water as an economic good is central to achieving equitable allocation and sustainable usage. Water allocations should be optimized by benefit and cost, and aim to maximize water benefits to society per unit cost. For example, low value uses could be reallocated to higher value uses such as basic drinking water supplies, if water quality permits. Similarly, lower quality water can be allocated to agricultural or industrial use (IWA, 2008).

Recently Merrett (2004) has developed a new analytic framework for the understanding of out-stream flows in any catchment or region. This is termed the area's hydrosocial balance because the flows from which it is constructed are hydrosocial rather than hydrological. This meta-theory covers both the flow quantities and quality.

Benefit in current climate

If balance between needs and resources, less periods of crisis (FR, Nixon, 2008).

Contribution to adaptation to climate change

[30] http://www.worldwaterforum5.org/

The increasing competition for the limited water resources in situations with floods and droughts indicates the effectiveness of IWRM implementation compared to the traditional sector approaches (SIC, 2009).

Potential conflicts and problems under changing climate, precaution level
Completely free hands for an easier adjustment (FR, Nixon, 2008).

Applicability
Universal but especially beneficial for areas under water stress, i.e. where the demand for water exceeds the available amount during a certain period or when poor quality restricts its use.

Example 23

M014. Measures to deal with abstraction pressures, e.g. abstraction licensing

Description of the problem[31]
The effects of abstracting (taking water from a water source) will vary depending on:

- Volume being abstracted
- Sensitivity of the ecosystem
- Seasonality
- Volumes returned
- Distances between abstraction and discharge points.

Uncontrolled abstraction in large quantities (e.g. for irrigation or industry) may cause crisis situations for water supply. From an environmental perspective over-abstraction of a water body may lead to:

- Reduced water flow
- Reduction of water resources
- Stress or mortality of fish and/or invertebrates
- Increased risk of pollution through reduced dilution
- Damage to our landscapes.

Description of the measure
Registration of abstraction structures is the first step in establishing a control over the exploitation of water resources. A further step is **water abstraction licencing**. In UK, for instance, a licence is needed for water abtractions more than 20 cubic metres of water per day from a river or stream, reservoir, lake, pond, canal, spring, or an underground source[32].
Presently, licences issued in UK are time limited (10 years) that need not be renewed if climate change reduces resource availability (Nixon, 2008).

Benefit in current climate

[31] http://www.ni-environment.gov.uk/water-home/water_resources/abstraction.htm

[32] http://www.environment-agency.gov.uk/business/topics/water/32032.aspx

Helps to improve access to water, to avoid use of crisis situation; can guide animation measures (e.g. by improving efficiency).

Contribution to adaptation to climate change
Used to guide political choices.

Potential problems under changing climate, precaution level
France described it as a win-win measure (Nixon, 2008).

Applicability
First of all in areas under water stress, i.e. where the demand for water exceeds the available amount during a certain period or when poor quality restricts its use.

5 Climate change adaptation strategies addressed specifically in the REFRESH Project

5.1 Management of riparian areas to control water temperature by the establishment of woody riparian vegetation along streams and rivers

Most of the solar energy within a landscape is dissipated by evaporation and condensation of water (cooling function; Ripl & Hildmann, 2000). This process dissipates energetic potentials in time and space. A much smaller amount of energy is dissipated by production (photosynthesis) and respiration (mineralisation) cycles of organic substances. Vegetation helps to smooth temperature or moisture gradients leaving less driving potentials for matter transport. Smoother gradients support high biodiversity, because of more niche space for species with different demands. Today's intensive landscape management has created extremely steep gradients, for example, between forest and field or forest and clearcut areas. Steep potentials accelerate matter transport and reduce biodiversity limiting the potential niche space and create much greater variability in time of the temperature and moisture distribution patterns.

Maintaining lower water temperatures will reduce the risk for dissolved oxygen depletion, increase the capacity of a stream to assimilate organic wastes, mitigates the impact of increasing thermal pollution, and may reduce in places the mortality of common fish and invertebrate species in mid-summer caused by temperature stress (Ghermandi et al., 2009). Although there is convincing evidence about the importance of the riparian zone in stream water temperature control, especially from forestry research, only in few cases (e.g., Twery & Hornbeck, 2001) the water goals have resulted in concrete management recommendations.

The riparian zone is the land area influenced by stream-derived moisture. Thus, the edge of a river is not its channel margin, but the edge of the riparian zone (Poole & Berman, 2001). Stream water temperature is determined by the interaction between the external drivers (such as solar radiation, air temperature, and windspeed) and the internal structure of the integrated stream system.

5.1.1 Temperature drivers

Water temperature is proportional to heat energy divided by the volume of water. In a modelling approach Caissie et al. (2001) noted that the empirical coefficient linking air to water temperatures was related to summer discharge. Therefore, stream temperature is dependent on both heat load and stream discharge; any process that influences heat load to the channel or discharge in the channel will influence channel water temperature and can be considered a driver of stream temperature (Poole & Berman, 2001).

Moore et al. (2005) described temperature change of a parcel of water that flows through a stream reach as a function of energy and water exchanges across the water surface and the streambed and banks by the following equation (see Figure 7):

$$\frac{dT_w}{dx} = \frac{\Sigma Q}{\rho C_p v D} + \frac{F_{gw}}{F}\left(T_{gw} - T_w\right) + \frac{F_{hyp}}{F}\left(T_{hyp} - T_w\right)$$

where dT_w/dx is the rate of change in the temperature (°C) of the water parcel with distance, x(m), as it flows downstream; ΣQ is the net heat exchange by radiation, turbulent exchange, and conduction across the water surface and bed (W/m^2); F is the streamflow (m^3/s); F_{gw} is the ground water inflow rate ($m^3/s/m$); F_{hyp} is the hyporheic exchange rate ($m^3/s/m$); T_{gw} and T_{hyp} are the ground water and hyporheic water temperatures, respectively (°C); ρ is the water density (kg/m^3); Cp is the specific heat of water (J/kg/°C); v is the local mean velocity (m/s); and D is the local mean depth (m).

Water temperatures can increase also during a fire from the intense heat of combustion or after a fire from increased solar radiation due to the loss of riparian vegetation (Pilliod et al., 2003).

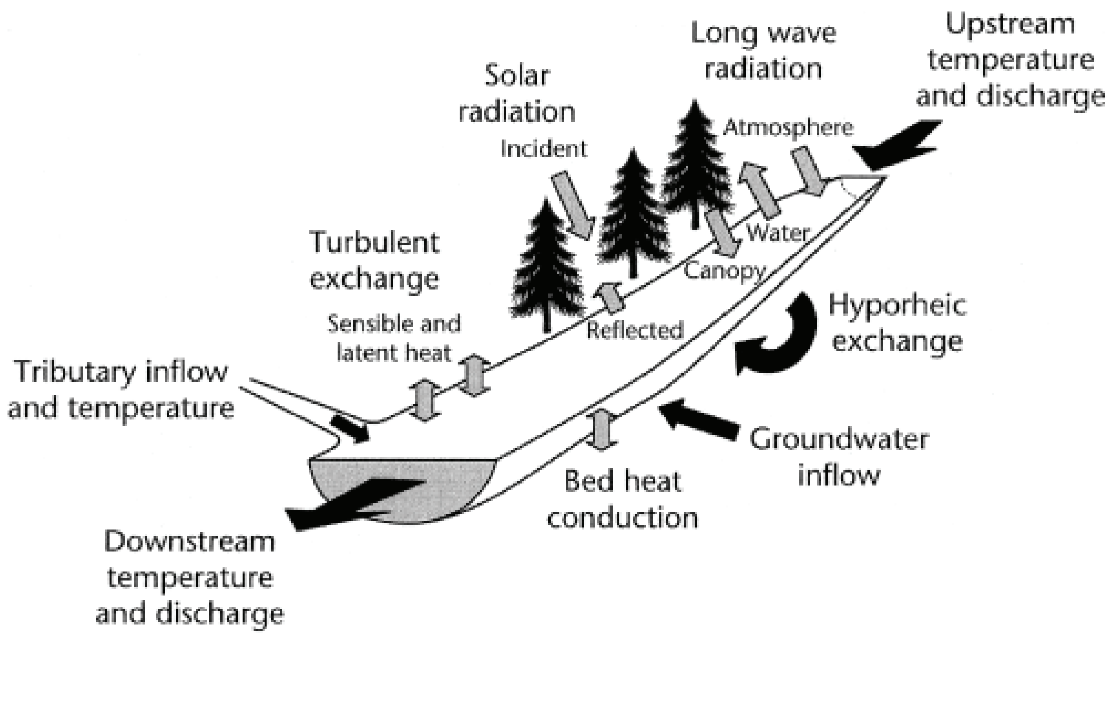


Fig. 7 *Factors controlling stream temperature. Energy fluxes associated with water exchanges are shown as black arrows (Moore et al., 2005)*

Main drivers affecting atmospheric heat load are air temperature, solar angle, cloud cover, topographic shade, upland vegetation, precipitation, wind speed, and relative humidity.

5.1.1.1 Shade

The radiation regime of streams is of major ecological importance because it controls light availability for photosynthesis and the thermal regime of the stream. Restoring riparian shade is often an important component of stream and riparian habitat rehabilitation (Poole & Berman, 2001; Davies-Colley & Rutherford, 2005). Although water temperature influences both chemical and biological processes, the water quality effects of riparian shading are largely unknown (Ghermandi et al., 2009).

Davies-Colley & Rutherford (2005) defined shade (*S*) as the complement of light exposure: $S=1-(I_i/I_0)$, where I_i is the light that reaches the site and I_0 is the incident light received at an open site. Shade under vegetation is measured by logging with light sensors or based on fisheye imaging of the canopy. Integration of LiDAR and QuickBird imagery can be used for mapping the distribution of overhang vegetation within the streambed and for measuring the width of both the riparian zone and the streambed (Arroyo et al., 2010).

Shade at the channel centre can be predicted by a simple model as a function of channel dimensions (stream width, *w*) and riparian plant character (foliage density, canopy height, *T*, see Fig. 8). The model reproduces the broad empirical trend of increasing shade with increasing *T/w* ratio (Davies-Colley & Rutherford, 2005).

Riparian forest buffer strips can also be dimensioned on the basis of the shading effect. To generalize the factors influencing the dynamics of shading ratio (S_n) at the water surface in watercourses, Mander (2008) presented the following equation based on the contributions of overstory vegetation to shading: $S_n = D/W^*\{(T^* \tan Z^* \sin|A-R|) - (Y - C/2)\}$ where *D* is the closeness of the canopy (%), *W* is the width of the watercourse (m), *T* is the height of the vegetation (m), *Z* is the angle between vertical plane and the line oriented to the Sun (in degrees; Fig. 8), *A-R* is the orientation of the stream stretch (in degrees), *Y* is the distance from the water table to the forest/bush (m), and *C* is the coefficient, depending on the form of tree crowns and their closeness.

Riparian vegetation may also play a role in protecting stream invertebrates from direct effects of UV radiation (Kelly et al., 2003).
Blocking solar radiation from reaching the channel, riparian vegetation reduces the stream's heat load. Vegetation also reduces near-stream wind speed and in this way the convectional and advectional heat exchange at the water-atmosphere interface (Poole & Berman, 2001). Riparian shade has a strong influence on temperature in small (1 and 2 order) streams (Anbumozhi et al. 2005), moderate influence in medium (3 and 4 order) streams, and the influence is low in large (5+ order) streams.

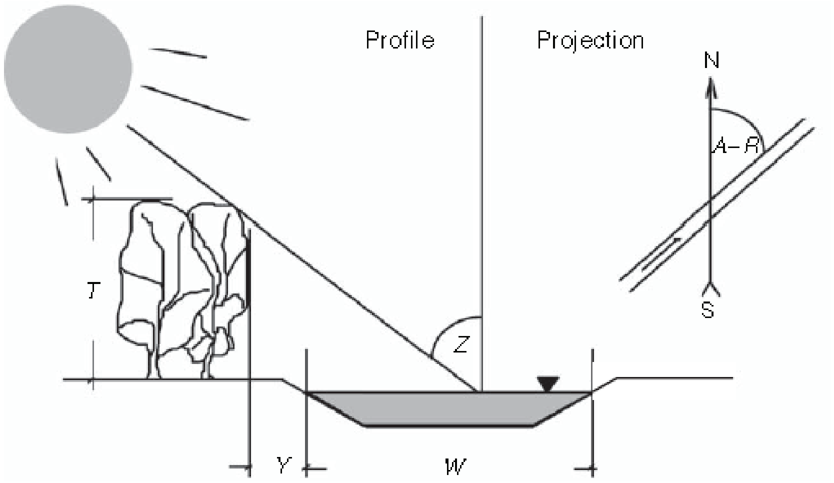


Fig. 8 *Scheme to estimate optimal parameters of streamside vegetation to stream surface shade*

Forest harvesting, particularly with removal of riparian vegetation, may result in stream heating or other changes in water temperature (Boothroyd et al., 2004; Moore et al., 2005) that could have deleterious effects on aquatic organisms. In their review Olson et al. (2007) showed that the magnitude of stream temperature response to harvest will vary with the amount of stream shade retained, the intensity of upslope harvest, and time since harvest. Complete removal of stream shade from headwater streams may result in temperature increases of as much as 5–13 ºC.

Stream temperature increases following forest harvesting are primarily controlled by changes in insulation but also depend on stream hydrology and channel morphology. Based upon the heat budgets of water mass in a small second order stream, Nakamura & Yamada (2005) showed that net solar radiation contributes the majority of energy and about 80% of the summer maximum water temperature variation can be explained by the sum of channel lengths without canopy cover.

Quinn & Wright-Stow (2008) analysed the influence of stream size on temperature impacts and recovery rates after clearfell logging. In many studies stream temperatures recovered to pre-harvest levels within 10 years but could take also longer. In papers reviewed by Olson et al. (2007), increased stream temperatures following harvest have been observed to persist from 5 years to more than 15 years.

Leaving riparian buffers can decrease the magnitude of stream temperature increases and changes to riparian microclimate, but substantial warming has been observed for streams within both unthinned and partial retention buffers. A range of studies has demonstrated that streams may or may not cool after flowing from clearings into shaded environments (Moore et al., 2005 and references therein), and further research is required in relation to the factors controlling downstream cooling.

5.1.1.2 Microclimate

Besides the direct effect of the riparian zone on water temperature, (reviews by Poole & Berman, 2001; Moore et al., 2005; Quinn & Wright-Stow, 2008) its role in creating a distinct microclimate has recently been extensively discussed (Moore et al., 2005; Olson et al., 2007; Brooks et al., 2009). The existence of a unique riparian zone, based on distinctive microclimate and floral and faunal communities has been repeatedly demonstrated for more arid, western conifer forests in the U.S. (Brooks et al., 2009). Brooks & Kyker-Snowman (2009) demonstrated the presence of a microclimatic riparian-upland gradient in western forests, with the stream effect extending out to 47 m for air and soil temperatures and 62 m for surface temperatures and humidity. The existence of similarly unique riparian zones in more mesic, eastern deciduous forests in the U.S. is uncertain, the research inconsistent.

Streams directly influence air temperature by acting as either a thermal sink (day, warm season) or source (night, cool season). In their review Moore et al. (2005) showed post-harvest increases in maximum summer temperatures ranging up to 13°C in most study streams in rain-dominated catchments. Summer daily temperature ranges after logging increased up to about 7-8°C, compared to pre-logging ranges of about 1-3°C. Surface and nearsurface soil temperatures show the largest differences between forest and open sites, being up to 10-15°C lower under forest canopies during the daytime and 1-2°C higher at night. Night air temperatures in forest areas are typically about 1°C higher than in the open landscapes. Studies in rain dominated catchments suggest that buffers may reduce but not entirely protect against increases in summer stream temperature caused by forest harvesting.

Near-surface water tables common to riparian areas indirectly influence microclimate by supporting development of vegetation and supplying moisture for transpiration from foliage (Olson et al., 2007). In forest stands, summer daily maximum air temperature tends to increase, and daily minimum relative humidity tends to decrease with distance from headwater streams. These effects appear more pronounced in non-maritime locations (inland from the coast). Trans-riparian microclimate gradients are typically non-linear with greater rates of change near-stream and smaller rates of change with distance upslope. Several studies (e.g., Anderson et al., 2007) reveal that the strongest influence of the air temperature gradient is expressed within approximately 10–15 m upslope from the stream. Generally the measured influence of streams on air temperature diminishes by distances of 30–60 m upslope of the stream in unharvested forests.

Within riparian buffers much of the change in microclimate takes place within about one tree height (15 to 60 m) of the edge (Moore et al., 2005). Solar radiation, wind speed, and soil temperature adjust to interior forest conditions more rapidly than do air temperature and relative humidity. In a study by Meleason & Quinn (2004) air temperatures were measured mid-way within a 5- and a 30-m wide riparian forest buffer and in an adjacent open clearfelled riparian area over an 11-month period. Median reductions in maximum daily air temperatures were 3.2°C in the 5-m wide and 3.4°C in the 30-m wide riparian forests as compared to maximum daily air temperatures in an adjacent treeless riparian area. Both forested sites had slightly warmer minimum daily air temperatures than those in the open site. These results suggest forested riparian buffers as narrow as 5-m wide can substantially moderate air temperatures as compared to a treeless environment. Wider buffers, however, will be needed for biotic connectivity. In a study within a headwater stream riparian zone in Oregon, USA, forest-floor invertebrate distributions were strongly associated with microclimate of riparian buffers of ~30-m wide that provided habitat for many riparian and forest species acting as effective forest refugia and dispersal corridors for invertebrates and other taxa (Rykken et al., 2007). Incorporation of riparian zones as into watershed management plans likely will contribute to meeting persistence and connectivity objectives.

5.1.1.3 Hydrological drivers

Hydrological drivers of stream temperature include groundwater temperature and discharge and the temperature and flow of tributaries (Poole & Berman, 2001). Although some streams in arid regions are fed only by surface runoff, most streams derive the majority of their discharge from groundwater. Depending on season, advection of ground water is an important source or sink for heat in ground water-fed streams (O'Driscoll & DeWalle, 2006). The temperature of the phreatic aquifer is generally the baseline temperature from which stream temperature deviates (although, as shown by Poole & Berman (2001) streams fed by snowfields and glaciers are exceptions to this rule). Removal of riparian vegetation can destabilize streambanks, thereby facilitating erosion, increasing sediment loads, and ultimately changing the physical structure of the stream hyporheic flow by reducing streambed permeability (Poole & Berman, 2001). Restoration of streambank vegetation likely will not be sufficient to meet temperature goals in streams where degraded channel morphology is the largest cause of undesirable stream temperatures.

5.1.1.4 Stream structure

While drivers determine heat and water delivery to the stream, stream structure determines stream channel resistance to warming or cooling. Loss of riparian vegetation may have major consequences for in-channel processes for forested streams since riparian vegetation is the primary source of large wood to the channel. The size of large wood and rate of large wood recruitment determine its influence on the channel; therefore current land-use practices such as the selective removal of standing riparian vegetation may have important ramifications for channel morphology (and therefore channel temperature) over time (Poole & Berman, 2001).

5.1.2 Human influence & management

5.1.2.1 Goals

Riparian management may have different water related goals, such as controlling runoff (Twery & Hornbeck, 2001), water quality (Mander, 2005; 2008), bank erosion (Ensign & Mallin, 2001), in-stream plant growth (Ghermandi et al., 2009;

Köhler et al., 2010), and creating habitats for warm- and cold-water fishes (Hendry et al., 2003).

Besides lowering the water temperature, shading reduces the light intensity at the water surface. Light penetrating the water column and reaching the bed controls, together with nutrient availability, hydraulic mixing, and grazing, the growth of algae and macrophytes. Light attenuation by plant canopy is likely the main determinant of the observed smaller periphyton biomass (Boothroyd et al., 2004; Ensign & Mallin, 2001) and macrophytes growth (Köhler et al., 2000, 2010; Mander, 1995) in shaded streams with respect to unshaded control sites. Mander (2005) found significant correlation (R^2 = 0.89; $p < 0.001$) between the shading rate of the stream surface and the biomass of aquatic macrophytes in lowland ditches of agricultural landscape in Estonia. Biomass below ecologically recommendable level causes disturbances in stream benthos ecosystem, particularly decreasing the biodiversity. Macrophyte biomass above the technologically allowable limit creates significant obstacles to water runoff. Bank vegetation at a shaded site of a temperate eutrophic lowland river decreased the light supply for macrophytes by 79%, the water column by 45% and the epiphyton by 28% during the vegetation period (Köhler et al., 2010). Growth of submersed macrophytes, but not of epiphyton, was light-limited in the shaded sections. Modelling of phytoplankton biomass in the River Ouse, N-E England, showed that reducing nutrient pollution would be less effective at suppressing the phytoplankton growth projected by climate change than the less costly option of establishing riparian shading (Hutchins et al., 2010).

Shade can be vitally important in shallow salmonid nursery areas, particularly during low flows when strong sunshine can cause summer water temperatures to rise to dangerously high levels for salmonids (Hendry et al., 2003). However, also the management of the riparian zone to prevent overshading was found critical (Hendry et al., 2003) in order to optimise the standing crop of salmonids. Removing shade would enhance habitats for warmwater fish.

5.1.2.2 Measures

Perhaps because of the widespread use of quantitative models (and associated simplifying assumptions), management actions seldom consider the multitude of interacting environmental processes that determine stream temperature regimes or the wide variety of pathways by which humans may affect stream temperature. According to Poole & Berman (2001) the key management implications of human influence on channel-water temperature include:

1) riparian vegetation structure (removal of upland or riparian vegetation)
2) modification of in-channel water flow that is a critical element for re-establishing desirable thermal regimes in streams (Dams);
3) alteration of groundwater dynamics (water withdrawals);
4) alteration of channel morphology (straightening, bank hardening, diking, etc.).

Retaining buffers of undisturbed riparian vegetation in logging areas can potentially reduce the magnitude of thermal disturbance to the stream from logging by maintaining a high level of stream shading (Quinn et al., 2004).

Riparian buffer width is typically dependent on the stream size to allow vegetation to stabilize banks, minimize the erosion of fine sediments into the channel, and provide shading which maintains the moderated microclimate (Anderson et al., 2007) and stream productivity

There is no consensus regarding the most effective buffer width to protect stream and riparian ecosystems. Depending on management goals, the following delineations for riparian management zone are typical (Olson et al., 2007):

~10 m for retaining stream bank stability to reduce sedimentation;

~15–30 m for maintaining instream habitat attributes such as water temperature, litter and wood inputs;

~40– 100 m for a more conservative approach for provision of instream habitat conditions with benefits to riparian-dependent species.

In the guidelines for the retention of treed riparian buffers after timber harvest in Canada and the United States, the mean buffer widths varied from 15 to 29 m for different waterbody types (Lee et al., 2004). Most common modifying factors in the guidelines were the waterbody type, shoreline slope, waterbody size, and presence of fish. In the United States protective regulations typically use an arbitrary distance from a stream, or a minimum distance that is extended based on slope (Brooks et al., 2009). Riparian zones have been defined by the plant community composition (Hagan et al., 2006) or amphibians (Perkins & Hunter, 2006). A 93 m wide riparian buffer zone was recommended to protect stream-breeding salamanders in southern Appalachian headwater streams (Crawford & Semlitsch, 2007).

Wilkerson et al. (2006) who evaluated the effect of timber harvesting on summer water temperature in first-order headwater streams in western Maine concluded that water temperature in these streams was protected from the effects of clearcutting by an 11-m buffer with 60% canopy retention. Streams without a buffer showed the greatest increase in mean weekly maximum temperatures following harvesting (1.4–4.4°C). Streams with an 11-m buffer showed minor, but not significant, increases (1.0–1.4°C). Streams with a 23-m buffer, partial-harvest treatment, and control streams showed no changes following harvest. The mean daily temperature fluctuations for streams without buffers increased from 1.5°C/day to 3.8°C/day, while with 11-m buffers fluctuations increased nonsignificantly by 0.5–0.7°C/day. Water temperatures 100 m below the harvest zone in the no-buffer treatment were elevated above preharvest levels.

At 28 sites in New Zealand studied by Boothroyd et al. (2004), the summer mean water temperature was greatest at the clearcut site (18.7 ºC) compared to pre-harvest (16.0 ºC) and post-harvest sites (16.6 ºC) with native buffers. Diurnal stream temperature ranges at pre- and postharvest sites with riparian buffers were lower (typically 3–4 ºC) than clearcut sites (up to 10–12 ºC). Water temperatures measured during the summer field surveys were 3.2 ºC higher at clearcut sites (mean 20.0 ºC) than at the pine and native forest sites.

Following tree planting in a degraded New Zealand pastoral stream, all planting scenarios predicted to decrease daily maximum water temperature after 15-20 years to levels that would be suitable for sensitive invertebrate species (Collier et al., 2001).

The following measures regarding riparian zone management were included into the database:

M380 - Creating riparian shading at small and moderate-size watercourses to control excessive algal growth during summer periods (Hutchins et al., 2010).

M388 - Fencing and the protection of riparian vegetation (Twery & Hornbeck, 2001).

M395 - Enhancing habitat for warmwater fish by (1) maintaining buffer strips; (2) removing some trees to allow more sunlight to reach the water surface; (3) allowing a few mature trees to die in place; and (4) prohibiting livestock from entering the buffer strip (Twery & Hornbeck, 2001).

M396 - Treatments for enhancing cold-water fish habitat by (1) maintaining buffer strips with at least 70% relative density; (2) allowing a few mature trees to die in place; and (3) creating small penings less than 0.1 ha in size (Twery & Hornbeck, 2001).

5.2 Management of catchment hydrology to maintain flow in streams, water-level in lakes and regular flooding in wetlands

5.2.1 Maintaining stream flow

Stream flows resulting from the complex hydrological balance of the landscape undergo naturally large seasonal and inter-annual changes. During low flows that occur in summer and in soil frost areas also in winter, water is normally derived from groundwater discharge or surface discharge from lakes, marshes, or melting glaciers. Low flow periods are critical considering the MUFS (Multiple Use and Functions of Water Services) and the river ecosystems are most vulnerable in these periods both to human impact and climate change.

5.2.1.1 Anthropogenic impacts on low flows

The principal activities by which flow regimes can be modified include land-use changes such as drainage and agricultural/forestry practices, water abstraction and transfer between catchments, impoundment and flow regulation, and hydro-electric power generation. In terms of simple hydrology, the effects of these various activities can be to reduce flow, to increase flow, and to modify patterns of flow fluctuations. They can be divided into activities affecting flow generating processes and those affecting directly the streamflow (Smakhtin, 2001). **Human activities affecting flow generating processes include:**

- groundwater abstraction;
- floodplain drainage for agricultural or construction purposes;

- changes of the vegetation in valley bottom areas through clearing or planting that modify the evapotranspiration loss from riparian soils;
- afforestation of a whole catchment or its parts;
- clearfelling and timber harvesting that increase annual water yield in many cases due to increase in seasonal low flows;
- catchment urbanisation.

Impacts directly on low streamflow include:
- direct river abstractions for industrial, agricultural or municipal purposes;
- direct effluent flows into river channels from industrial or municipal sources;
- irrigation return flows from agricultural fields;
- direct importation of water from outside the catchment via inter-basin transfer schemes;
- construction of dams and subsequent regulation of a river flow regime.

5.2.1.2 Environmental impacts of instream flow reduction

Low flow periods may result in:
- increased sedimentation that changes the morphology of the stream channel and flood plain,
- changes in distribution and abundance of stream biota,
- aggravation of the effects of water pollution due to reduced dilution capacity,
- increase in water temperatures.

Caruso (2002) described the following effects of summer extreme low flows in Otago Region, New Zealand in 1998-99:
- At most sites temperature was slightly higher during longer period.
- Bacterial contamination occurred in pastoral catchments due to increased livestock use of watercourses and decreased dilution.
- Concentrations of N, P and SS decreased due to lack of runoff events.
- Conductivity increased due to evaporation.
- Diversity of macroinvertebrates decreased slightly.

However, water quality and biodiversity recovered rapidly in autumn indicating resilience of the streams to extreme low flows.

Lowering the water table and/or reducing overbank flooding may result also in changes in the density, productivity, and species composition of wetland and riparian vegetation.

5.2.1.3 Management options

Recognising the mechanisms of anthropogenic and climatic impacts enables to find management measures to revert them. Management of rivers for some specific purpose (e.g., to satisfy fish requirements) is no longer viewed as an entirely valid approach (Smakhtin, 2001). Rivers should be considered as balanced ecosystems and recommendations are often required as to instream flows which would ensure fish passage, temperature levels, different habitats maintenance, sedimentation control, recreation etc. With the increasing pressure on water resources it should be recognised that aquatic ecosystems are not just equal competing water users among the others, but the base of the resource itself, which needs active care for a sustainable development.

The influence of land management on hydrology is governed by a number of complex processes and factors such as soil type, gradient, nature of drainage and scale are all important. Hence there are dangers in over-simplification of cause and effect (Hendry et al., 2003). The complex character of hydrological interrelationships in soils can be exemplified by the fact that drainage reduces peak flows from clay soils but increases them from permeable soils (Burt, 1995). Farming practices like tillage affect soil compaction and may have a significant effect on flows under local circumstances but their significance at the catchment scale, particularly in larger systems remains unclear. Upland drainage has been shown to influence the hydrology of rivers, resulting in increases in peak flows and increased velocities, the combined effect of agricultural under-drainage and channel improvements is to generate quicker, higher peak flows but not to alter flood volumes. The main principles of river restoration should include the establishment of sustainable and environmentally acceptable land practices in the catchment and ensure that any management solutions adopted would be tailored to the local needs wherever possible restoring the natural flow regime (Hendry et al., 2003).

Forestry operations such as afforestation, harvesting, and road construction can have a significant impact on hydrology at the site, hillslope, and catchment scales. At the catchment scale, harvest may decrease transpiration and result in a transitory period of higher summer minimum flows lasting from a few years to more than a decade. Peak flows increased following forest harvesting in most studies in coastal catchments, with increases ranging from 13 percent to over 40 percent (Moore & Wondzell, 2005).

Most rain falling onto a canopy will be intercepted by foliage, and some of that amount will be lost to the atmosphere by evaporation. On an annual or seasonal basis, interception loss from conifer forests in the U.S. Pacific Northwest generally represents about 10 to 30 percent of total rainfall, depending on canopy characteristics and climatic conditions (Moore & Wondzell, 2005). Decreased interception loss would increase the amount of water infiltrating the soil, leading to higher water table levels during storms.

If the goal of water management is to **increase water yields** to streams and reservoirs during periods of low flows, the following two measures can be applied:

M391 Clearcutting to increase water yield to streams and reservoirs during periods of low flows.

M392 Silvicultural measures to protect against reductions in low flow levels during the next rotation ((1) reducing stand stocking to below 70% relative density; (2) using short cutting cycles; (3) using short rotations; (4) encouraging hardwood species; (5) encouraging regeneration from seedlings rather than sprouts; (6) avoiding conversion to softwood species from hardwood species. Changes in species, such as converting hardwoods to softwoods, can reduce water yield, especially during periods of low flow (Twery & Hornbeck, 2001).

Increases during low flow periods will lead to greater overall water yield and also may increase the magnitude of high-frequency flood events. This occurs because soil water content in recently harvested areas is higher than in fully stocked stands. The treatment effect tends to be short-lived and as storm size or snowmelt volume increases, this difference is less important. In the absence of other management goals, the recommended silvicultural systems will provide optimum reduction in evapotranspiration. Generally this choice will be short-rotation, even-age silviculture with clearcutting as the harvest method. Changes in species, such as converting hardwoods to softwoods, can reduce water yield, especially during periods of low flow (Twery & Hornbeck, 2001).

It is important to mention that in certain areas forest stands may have an **opposite function**. Fog or clouds can be intercepted by trees, condense onto the canopy, and then drip to the forest floor as **interception gain**. Fog and cloud drip may significantly augment precipitation in coastal catchments and in some high mountain catchments where forest harvesting may actually reduce streamflow (Moore & Wondzell, 2005). Reducing forest loggings and afforestation in coastal areas could be effective measures against desertification contributing to more homogenous distribution of precipitation:

M401	Reduce forest loggings in coastal and high mountain catchments to collect fog and cloud drip as canopy interception gain (Moore & Wondzell, 2005)

Afforestation (**M029**, see Example 2) may cause a significant reduction in total flow. Cheng et al. (2002) evaluated the hydrologic influences of forests in Taiwan. Despite rainfall intensities often exceeding 100 mm/h, overland flow rarely occurs on Taiwan's permeable forest soils. High evapotranspiration of 800-1200 mm annually contributes to reduce streamflows. Maintaining intact forest could reduce flood risk and contribute to the shortening of the water cycle:

M393	Silvicultural measures to limit peak (flood) flows by (1) maximizing stand stocking and maintaining it above 70% relative density; (2) using long cutting cycles; (3) using long rotations; (4) encouraging conifer species; and maintaining less than 25% of the area in openings or trees less than 10 years old (Twery & Hornbeck, 2001)

Several studies have demonstrated (either by field experiments or by modelling) that afforestation has had a major effect on low flows reducing low-flow volumes to a larger degree than those of annual flow (Moore & Wondzell, 2005). Hence the flood abating effect of this measure should be compared with the potential harm caused by decreased low flows. Afforestation of upland catchments with fast growing plantations can have significant impact on *in situ* water use, with consequent impacts on water availability downstream (Trabucco et al., 2008).

There are several other effects of forestry on hydrologic balance and runoff formation within a catchment (Moore & Wondzell, 2005). **Snow accumulation** tends to be higher in openings than under forest canopies, with cut blocks typically accumulating about 30 percent to 50 percent more snow. **Snowmelt rates** are typically 30 percent higher in the open landscape than in forest. During mid-winter snowmelt caused by rain-on-snow, condensation of water vapour by sensible heat transfer from the relatively warm air onto the snowpack, may significantly augment rainfall, increasing the magnitude of flood peaks.

Changes in infiltration rate created by forestry operations can increase overland flow. Undisturbed forest soils normally have sufficiently high hydraulic conductivities minimizing production of overland flow. Removal of the organic horizons diminishes infiltration via macropores and rain splash, especially on recently disturbed soils, can detach fines that can clog soil pores. At many sites, harvesting is conducted with skidders, tractors, or other ground-based equipment, which can cause compaction of the soil surface to depths of 30 cm or more. Excavated trails and constructed haul roads typically have compact surfaces with low permeability and can generate infiltration excess overland flow in even moderate rainstorms (Moore & Wondzell, 2005). Afforestation could be a measure to restore the hydrologic function to previously disturbed lands:

M395	Silvicultural measures to restore hydrologic function to previously disturbed lands by maximizing leaf litter and coarse woody debris, and slowing the decomposition rate (Twery & Hornbeck, 2001).

Subsurface stormflow is a dominant process in undisturbed forested catchments. Forest operations can influence subsurface stormflow in several ways. Logging roads can intercept shallow subsurface flow and lead it via ditches and culverts more rapidly to the stream network, potentially leading to **increased peak flows**. Forest operations can influence channel characteristics either directly by removal of wood or loading of slash into the stream or indirectly via logging related debris flows (Moore & Wondzell, 2005).

Urbanisation in some forested watersheds may cause increased peak flows and decreased low flows due to significantly reduced soil infiltration capacities (e.g., Cheng et al., 2002).

River regulation. A general feature of river reservoirs constructed for direct supply, river regulation, hydropower generation or a combination of these uses, is that they release compensation water to the impounded river. The pattern of these releases varies, being either set at a fixed daily volume or with some seasonally based variation. The quantities also vary, historically often based on 1/8th average daily flow of the impounded stream, or more recently the Q95 (the flow rate that is exceeded for 95% of the time).

In some cases the compensation flow provides a significant benefit over the natural flows experienced in dry and drought years. Reservoirs offer the potential to control the timing and volume of releases, and any adverse environmental impacts of a reservoir and its operation may in part be offset through innovative management of releases. Reservoirs offer the opportunity to provide flushing flows before the spawning season to rework the gravels and remove the silt deposits (Hendry et al., 2003). Based on this information, the following measures can be formulated:

M404	Maintain compensation flows at dam reservoirs.

M405 Releases from dam reservoirs to stimulate upstream migration of adult salmon and sea trout or downstream migration of smolts, to provide flushing flows to clean gravels prior to the spawning season.

M406 Adjust release patterns from hydroelectric power plants to simulate a natural spate hydrograph for the river.

If the river bed below a dam reservoir is too wide to allow enough flow formation in the channel for fish migration, further hydromorphological modification of the channel may be necessary, e.g.:

M407 Designing a two-stage channel to reinstate favourable velocity and depth conditions below a dam reservoir. The low flow channel will carry compensation flows and normal regulation releases, and a higher level channel over shallow side berms will carry high flows (Hendry et al., 2003) .

Water resource and flood alleviation schemes can be used to great advantage in overcoming water quality problems commonly experienced during low flows in estuaries. For example, in the highly regulated Welsh Dee catchment, a water bank is available for specific fisheries purposes to allow quantities of water to be released to improve quality in the estuary by dilution. Even under naturally low summer flows, this somewhat artificial intervention can provide a much needed refuge area for migratory salmonids trapped in otherwise poor water quality conditions in the estuary (Hendry et al., 2003)

5.2.2 Maintaining regular flooding of wetlands

Flooding regime of wetlands is closely related to river flow as regular overbank flooding is vital for maintaining the health of many riparian ecosystems. Periodic flooding controls the plant community by disturbance favouring pioneer species (Azami et al., 2004). Reduced floodings due to hydrological modifications can allow further succession of plant communities beyond a typical stage (Goodwin et al., 1997; Azami et al., 2004).

Changed flood regimes can favour exotic species. Bhattacharjee et al. (2009) have described a case in the southwest U.S. where suppressed annual flooding cycles has supported the dominance of saltcedar (*Tamarix chinensis*) over the native cottonwood (*Populus deltoides*). In conditions of periodic floodings, cottonwoods were competitively superior to saltcedar but the lack of floodings created more favourable conditions for the germination of saltcedar.

When altered flow regimes have impacted riparian zone health, re-establishing natural streamflow is the first step to effectively restore riparian ecosystems.

Many of the eco-hydrological functions of floodplains are strongly related to the interactions between the floodplains shallow groundwater and the surface water of the usually well connected lowland rivers (Butturini et al., 2002; Hancock, 2002; Hancock et al., 2005). The interactions between groundwater and surface water and the resulting exchange fluxes are often characterised by a high temporal and spatial variability (Krause et al., 2007). Commonly the type of interaction is described by the direction of the exchange fluxes distinguishing between influent (flowing in) fluxes and effluent (flowing out) fluxes. Based on these fluxes the investigated streams/stream reaches are described as loosing, gaining and through flow or parallel flow (Sophocleous, 2002). Reductions in groundwater abstraction may be needed to restore riparian ecosystems by reestablishing groundwater levels; however, groundwater withdrawal regulations do not usually incorporate provisions for riparian protection (Stomberg et al., 1996).

5.2.3 Maintaining lake water levels

Maintaining water levels in lakes may have totally different meaning and, consequently, should be governed by different measures in affluent areas and in areas of water scarcity. In water rich areas, especially if an increase in precipitation amounts or changes in its seasonal distribution are anticipated, the main challenges are related with avoiding floods and erosion flows. In water scarce regions the question is rather how to avoid shrinking of lakes and increase of salinity and pollutant levels, maintaining at the same time flow in the effluent river (if the lake is exorheic).

5.2.3.1 Avoiding lake floodings in affluent regions

Hydrological engineering projects, with broadly different aims and variable degrees of intensity, are currently affecting practically all large water bodies in Europe. In many cases, and especially in lowland lakes, the protection of economic interests demands control of the natural flooding of the system (van den Brink et al., 2005). Generally, the opposite case holds for hydropower: electricity production mostly calls for unnatural water level alterations and major hydromorphological modifications in the regulated basin. Various engineering operations are currently planned or pursued to protect the assets of lakes against extremes of weather or climatic change. Obvious risk areas are coastal lagoons and impoundments, such as the Dutch IJsselmeer (e.g., Breukers, 2000), for which even a small rise in oceanic level will have serious consequences.

In northern Europe increased melting of snow and rainfalls in projected milder winters (Räisänen et al., 2004) will increase winter floods. Spring floods will be reduced when the snow cover will no longer accumulate during warm winters. Water levels in the large lakes in central Finland in winter will become higher than now. An increase in summer floods, particularly in small lakes, is expected as a result of the wider spread of torrential rains during summer. On the other hand, a prolonged summer season also brings the opportunity for dry summers, particularly in southern and central Finland. According to the RBMPs for Finland (Kotanen et al., 2009), growing winter runoff and more frequent winter floods of southern and central Finland require leaving more storage capacity in the regulated lakes. In spring, the need for storage capacity will be lower when the snowmelt floods will disappear or be reduced. Because of longer and sometimes also drier summers, the lakes need to be filled in spring. In northern Finland, the storage capacity is still needed to reduce the flood risk caused by snowmelt. Lake regulation permits need to be changed. The need for change is estimated for more than half of the current 220 regulation permits. Based on the Finnish example, the following management measures for the regulated lakes could be formulated:

M264 Leave more storage capacity for winter in the regulated lakes in southern and central Finland because of increasing winter runoff and more frequent winter floods.

M265 Need for lower storage capacity in spring when the snowmelt floods will disappear or be reduced.

M266 Need to fill the lakes in spring because of longer and sometimes also drier summers.

M267 Continuing need for the storage capacity in northern Finland to reduce the flood risk caused by snowmelt.

M268 Need to change more than half of the current 220 lake regulation permits.

5.2.3.2 Avoiding water level decrease in areas of water shortage

Another sensitive basin type is represented by lake basins of arid climate zones, for which even subtle changes in the precipitation–evaporation balance may prove fatal. Such lakes are often additionally affected by water abstraction, e.g., for irrigation. The huge, high-elevation Lake Van (3,700 km^2, max depth 450 m, elevation 1,719 m ASL) is a rather extreme example of an endorheic alkaline lake, for which major changes may be anticipated as a consequence of increased evaporation due to climatic change (Altunkaynak et al., 2003). The lake’s paleolimnological sediment record indicates major level fluctuations connected with events during the climatic history of the Holocene (Wick et al., 2003).

Over-exploitation of the water resources of Lake Sevan caused a severe water level reduction of 19.2 m, which reduced water volume by 42.2%, decreased the hypolimnion volume by 90% and caused water quality degradation (Hovhanissian & Gabrielyan, 2000).

As an extreme example, the Aral Sea, a landlocked saline lake shared by Uzbekistan and Kazakhstan in Central Asia, has undergone dramatic shrinking and increased salinization since the 1960's (Alekseeva et al., 2009). The waters of the Amu-Darya and Syr-Darya rivers, which previously supplied almost 90% of the Aral Sea freshwater inflows (Khan et al., 2004), have been diverted for irrigation. Due to decreasing river discharge, evaporative losses have become much larger than the freshwater inflows in the water budget of the Aral Sea. Consequently, the sea level has dropped by almost 23 m (Zavialov et al., 2003) since the 1960's and the surface area of the Aral Sea has decreased to one-third of its former size. In the 1990's, it eventually split into two seas, the Large Aral and the Small Aral. Moreover, the continuous evaporative loss resulted in extreme salinization.

To reduce evaporation losses, in 2005, the Kokaral dike was built across a narrow stretch of the Aral Sea, splitting off the North Aral Sea (also called "The Small Sea") from the much larger South Aral Sea ("The Large Sea"). This extreme measure was aiming at restoring the water levels in the northern part of the lake while leaving the rest dry out. Alternatively water level would have continued dropping and the salinity increasing in the whole lake. Water level of the North Aral has risen, and its salinity has decreased. Recovery of sea level has been more rapid than expected. The dam has caused the small Aral's sea level to rise swiftly from less than 30 m to 38 m, with 42 m considered the level of viability (Cretaux et al., 2005).

5.3 Re-creation of riparian floodplains to buffer against extreme precipitation events and changes in hydrodynamics, and to reduce flows of nutrients and humic substances to water bodies

5.3.1 Re-creation of riparian floodplains

Degradation of riparian zones is almost a world-wide phenomenon (Richardson et al., 2007). The recognition of the high biodiversity of riparian zones as habitat and wildlife corridors, and their important role in flood and erosion control, water provisioning and protecting of water quality has led to numerous restoration projects of riparian floodplains and their ecosystems (Goodwin et al., 1997; Bissels et al., 2004; Aguilar-Ibarra et al., 2005; Somper, 2005; Richardson et al., 2007; Moss & Monstadt, 2008; Ducrotoy & Dauvin, 2008). In most cases, restoration is guided by an understanding of riparian zone ecological processes and causes of degradation (Goodwin et al., 1997). The efforts are often made simultaneously with stream restoration projects.

5.3.1.1 Causes of riparian zone degradation

World-wide the most part of wetland loss and degradation is caused by

- drainage for agriculture,
- infrastructure developments,
- afforestation and malaria control,
- blocking and extraction of the water inflow,
- over-exploitation of groundwater resources,
- building of dams,
- pollution from agricultural and industrial sources.

According to the EEA report on biodiversity (EEA, 2010), in 2006, the total area of wetland ecosystems in the EU was over 79 000 km^2 comprising watercourses and water bodies (45%), marshes and bogs (32%), and coastal wetlands (23%).

- In 2006, the total area of wetland ecosystems was around 0.5% smaller than in 1990 for the same geographical area.
- In 2006, for the same geographical area as surveyed in 1990, watercourses and water bodies had increased by 4.4%; marshes and bogs had decreased by 5.0%; and coastal wetlands had increased by 0.4%.

Between 1990 and 2006, 394 km^2 of wetland ecosystems was lost and 1187 km^2 of wetlands excluding water bodies was lost. The highest rate of loss (– 5.0%) was to marshes and bogs, but the rate of loss was lower between 2000 and 2006 (– 0.9%). Some 35% of the change in wetland areas between 2000 and 2006 was due to conversion to agriculture (Fig. 9) and 49% to forest creation and afforestation. Of the wetland area converted to other land uses between 1990 and 2000, 2% were artificialised (e.g., turned into urban areas), 7% became agricultural, 12% water bodies, and 79% forest and semi-natural areas.

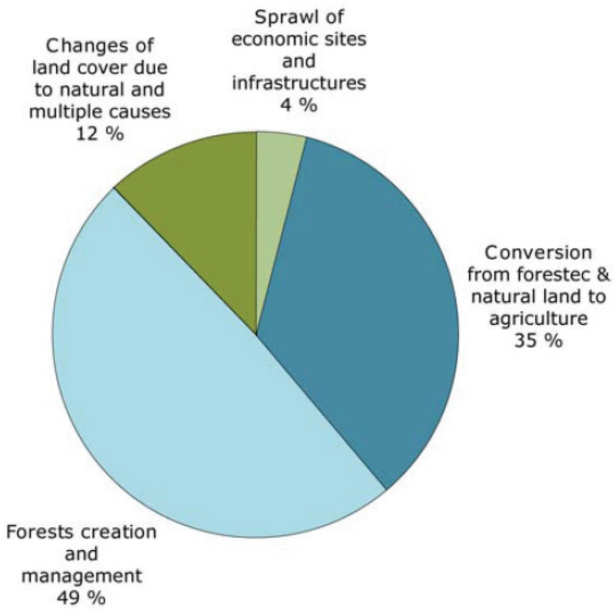


Fig. 9 *Cause of loss of wetland ecosystems in EU (except Greece and the United Kingdom) between 2000 and 2006 (EEA, 2010)*

Whatever the specific causes, riparian zone disturbances can be divided into:

- **hydrological modifications** such as drainade, dams and diversions, stream and groundwater abstraction, stream channelization and levee construction that change stream morphology and hydrological processes,
- **habitat alterations** such as land clearing, livestock grazing, mining, and invasions of alien species that result in direct modification of riparian communities.

5.3.1.2 Goals of restoration

Besides reconstructing a riparian area to correspond as closely as possible to the reference conditions, slightly different aims can be considered (**M030-M032**, Example 4). For instance, different procedures of wetland management applied by the U.S. Natural Resources Conservation Service[33] aim at:

- inundation of lands to provide habitat for fish and/or wildlife,
- providing watering places for wildlife,
- rehabilitation of degraded wetlands,
- modification of an existing wetland where specific attributes are targeted by management objectives,
- treatment of point and non-point sources of water pollution,
- creating a wetland on a site which historically was not a wetland.

5.3.1.3 Methods of riparian zone restoration

Methods for restoring riparian zones are often determined by the cause of degradation. Two main steps in riparian zone restoration are (1) restoring geomorphic features and hydrological buffer function, and (2) re-establishing riparian vegetation and nutrient/erosion buffer function.

5.3.1.3.1 Restoring buffer zone hydrology

Well maintained riparian wetlands offer wash lands which can store flood water for later controlled release. Regardless of the cause or nature of wetland degradation, the goal of restoration is often to return the degraded site to as near its original state as possible, in terms of both ecological function and habitat for native flora and fauna (**M030-032**; **M038; M295**). Restoration of the ability to store water (**M295**) is the first priority.

When altered flow regimes have degraded the riparian zone, re-establishing natural streamflow may be the most effective restoration measure for riparian ecosystems. If the flow regime has been altered by a dam and its removal (**M004; M356; M386**) is not feasible, maintaining compensation flows at dam reservoirs (**M404**) and adjusting release patterns from dams plants to simulate a natural spate hydrograph for the river (**M406**) can be good alternatives (Hendry et al., 2003). Moreover, controlled releases from dam reservoirs can stimulate upstream migration of adult salmon and sea trout or downstream migration of smolts, and provide flushing flows to clean gravels prior to the spawning season (**M405**).

Decreasing the drainage of groundwater by canals (**M009**) and preventing overexploitation of groundwater (**M232**) may also help restore riparian ecosystems by re-establishing groundwater levels that favour riparian vegetation. An administrative measure would be prohibition on issuing water abstraction permits in the aquifer areas of groundwater-dependent intermittent rivers (**M250**), however, this would require providing water supply from alternative sources (**M333**).

For planned inundations of protected (former natural) floodplains (**M025**), a physical restoration of the stream channel (**M352**) may be necessary. To restore instream benthic habitats and a narrow riparian buffer strip, it may be enough to

[33] http://www.nrcs.usda.gov/

move embankments further away from banks and shores reducing in this way pressures from hard engineering structures on beds, banks and shores (**M354**). For more robust rehabilitation, hydromorphological measures like meandering streams creating ecological designed banks (**M059; M352**) may be necessary. For restoration success, particularly if entirely new channels are created, restoration plans must take into account the geomorphic potential of the individual stream and tailor restoration methods accordingly (Rosgen, 1997). If he goal of geomorphic restoration is to eventually restore hydrologic processes important to riparian and instream ecosystems, stream channels should be designed narrow enough to overflow into the floodplain on a 1.5 to 2 year timescale (**M452** - Rosgen, 1997).

In peatlands water losses are reduced directly by blocking drains (**M116; M378**) or, more rarely, by installing waterproof membranes along the perimeter. On cutover peatlands, drains are sometimes filled with peat and the bare surface reshaped to create dams or bunds that reduce overland flow. Opportunities for such large-scale engineering are more limited on planted peatlands, because the surface vegetation is still intact.

5.3.1.3.2 Re-establishing riparian vegetation and buffer function

By restoring hydrological processes such as periodic flooding that favour riparian vegetation, native communities may regenerate on their own (Trowbridge, 2007). Planting of efforts may still be needed for fast establishment of riparian vegetation (Young et al., 2001). For example, re-establishing clonal species such as willows can be accomplished by placing bundles of live willow withies behind the trunks at stream shores where they will develop roots and shoots, binding bank material together and trapping silts which act as a growth medium for colonising emergent macrophytes (**M453** – Hemphill & Bramley, 1989) or simply putting cuttings directly into the ground (Stromberg, 1993).

The recommended width of a buffer is 10 m for upland streams and 100 m for lowland rivers (Hendry et al., 2003). Buffer plants such as *Phragmites communis* and *Typha latifola* can be used to create a riparian reedbed system which will act as an effective substrate–plant biofilter. These species are capable of high rates of growth associated with elevated levels of nutrient uptake and demand (particularly nitrogen and phosphorous) that makes them especially useful for capturing polluted runoff from steadings in constructed farm wetlands (**M335**). The presence of a wider land-based riparian plant community will also act as a physical barrier to pollutants, retarding their translocation from soil to water as well as having a role as a biofilter.

Recent restoration of planted peatland sites in northern Scotland included felling exotic trees and placing them intact into the ditches. The expectation was that branches of the felled trees will act as a climbing frame, enhancing growth of the peat-forming moss, Sphagnum, in the ditches (Belyea, 2004).

In logging areas retaining buffers of undisturbed riparian vegetation can potentially reduce the magnitude of disturbance to the stream from logging by filtering sediments and nutrients in runoff, reducing soil and vegetation disturbance in the near stream area, maintaining a high level of stream shading and leaf litter input, and reducing input of logging debris and associated streambank disturbance (Quinn et al., 2004).

Forestry measures for intensive protection of riparian areas (**M398**) include (1) maintaining buffer strips with at least 70% relative density; (2) allowing a few mature trees to die in place; and (3) creating small openings less than 0.1 ha in size (Twery & Hornbeck, 2001).

Vegetation of planted or maintained buffer strips may need to be protected from grazing. Fencing and protection of riparian vegetation (**M388**) by excluding livestock would prevent grazing of riparian vegetation promoting channel stability, controlling erosion, diversification of habitat by providing shade, and increased in-stream productivity by leaf litter. Preventing cattle grazing in riparian zones can allow riparian vegetation to rapidly increase in robustness and cover, and also shift to a more natural community composition (Dobkin et al., 1998; Sarr, 2002).

5.3.1.3.3 Nutrient retention

In rural watersheds the riparian land use remains a crucial link between the agricultural lands and the stream environment. It has been widely believed that these riparian buffer zones are effective in reducing the nutrient concentrations in water that pass through them. The first evidence for the role of riparian zones in buffering nutrient input from adjacent fields was provided by Peterjohn & Correll (1984); since then, numerous studies (e.g., McClain et al., 2003; Sabater et al., 2003; Mander et al., 2005; Mander, 2008) have evaluated the capacity of these channel-marginal wetlands to retain or remove nutrients and other pollutants. Riparian wetlands appear to be capable to improve water quality within streams.

For the point of view of watershed and landscape management, the riparian zone can be divided into two major functional parts

- the riparian buffer zone (**M067; M383; M398**) and
- the riparian buffer strip (**M083; M370; M395; M396**).

The first is wider (50–1500 m) and has less strict management prescriptions, whereas the buffer strips are narrow areas at the riverbanks and lakeshores with very limited management opportunities (Mander, 2008). In an ideal (undisturbed natural) case, the structure of riparian zones and strips can be coherent with the complexity of natural river corridors.

A general, multi-purpose, riparian buffer design consists of a strip of grass, shrubs, and trees between the normal bank-full water level and cropland (**M384**, Anbumozhi et al., 2005). The spacing of trees, shrubs, and grasses at different distances from the stream and the crop field is the primary objective of the design. However, the design considerations should also include easier maintenance activities, such as mowing or mulching, control burning, and other activities within the buffer strips. The design consideration should also balance the benefits, like stabilizing the bank, improving and protecting the aquatic environment, and protecting cropland from flood erosion and debris damage.

The ability of riparian vegetation to retain and/or remove nitrogen (Ensign & Mallin, 2001; Sabater et al., 2003; Anbumozhi et al., 2005; Dodds & Oakes, 2006; Hefting et al., 2005) and phosphorus inputs (Ensign & Mallin, 2001; Meals, 2001; Borin et al., 2004;) originating from upslope diffuse and point sources is well documented and generally acknowledged.

Mander et al. (2005) showed that removal of materials (suspended solids, nutrients, organic material, heavy metals, pesticides) has a non-linear character: in the first part of the buffer (0–5m from the field-buffer borderline), significantly more material (20–60%) is retained than in the remote parts of the buffering ecosystem. The removal process can be described by the following equation:

$$C_L = (1 - e^{-kL}) \times 100\%$$

where C_L is the change in concentration (%) at distance L (m) from the buffer boundary, k the removal rate coefficient (m^{-1}); $k=(\ln C_1 - \ln C_2)/L$, where C_1 is the initial concentration at the field buffer boundary and C_2 is the concentration at the distance L from the boundary. A strong linear regression has been found between logarithmic values of both N and P initial load (x) and mass removal (y) in buffer strips. However, the relative removal efficiency y/x for both N and P is decreasing when x increases.

The efficiency of riparian buffer zones and buffer strips in water purification can be described using three characteristics (Mander, 2008): removal efficiency (%), the retention capacity or mass removal (kg ha^{-1} yr^{-1}), and the specific removal (% m^{-1}).

Removal efficiency *E* (%) of N and P in riparian communities and constructed wetlands was estimated as

$$E = 100\% * (Q_{in}C_{in} - Q_{out}C_{out})/(Q_{in}C_{in})$$

Where Q_{in} and Q_{out} are inflow and outflow values (m^3 d^{-1}), respectively; C_{in} and C_{out} are concentration values (mg l^{-1}).

The retention capacity *R* (kg ha^{-1} yr^{-1}) was calculated as:

$$R = \sum(Q_{in}C_{in} - Q_{out}C_{out})/A$$

where $\sum(Q_{in}C_{in} - Q_{out}C_{out})$ is the annual retention and A is the area of the buffer zone.

The specific removal (% m^{-1}) is defined as the removal efficiency per unit width of a buffer zone. This characteristic is useful for the planning and establishment of buffer communities. The high retention efficiency of buffer strips depends mainly on the heterogeneity of the loading events, i.e., the best results occur when the polluted water from adjacent fields enters buffers in short events (e.g., during intensive rainfalls and/or intensive thaws).

To maximise the removal of nitrogen and phosphorous via direct uptake into the plant tissue, high growth rates and levels of standing biomass must be achieved. Hence frequent harvesting may be required to remove the accumulated nutrients, encourage new growth, and prevent any release of pollutants from senescent plant material (Hendry et al., 2003).

A combination of grasslands (wet meadows) as wider buffer zones (10–50 m) and forest/ bush communities as buffer strips (5–10 m) on stream banks is the most optimal structure of riparian buffer communities (Mander, 2008). The width of riparian buffer strips depends on the soil and relief conditions of the adjacent landscape, and normally lies between 5 and 50 m. This can be determined on the basis of maps of reclaimed areas at a scale of 1:2000 with detailed topographic and soil data, using the following formula applied in the planning of buffer zones

$$P = 0.000\,69 \frac{q \times f \times \sqrt{i}}{m \times K_i \times n}$$

in Estonia
where P is the optimal width of forest/bush buffer strip (m); q is the mean intensity of overland flow during the thawing period (mm d^{-1}; for Estonia q=8.4); f is the specific slope length (m); i is the mean slope in the catchment (i=tanα); m is the roughness coefficient of the surface in the catchment (mean value for ploughed fields: 1.0, for intensively managed grasslands: 1.1, for natural meadows: 1.2); K_i is the water infiltration within the buffer strip during the spring (mm min^{-1}; mean value over different soil types normally varies between 0.1 and 1.0); n is the soil absorption capacity; and the constant 0.00069 is the time variation coefficient (from days to minutes).

The mass removal of nitrogen and phosphorus in buffer zones can be negative when the input value is lower than a certain threshold (e.g., <0.3 mgN l^{-1}). On the other hand, the purification efficiency was always positive when the input value exceeded a certain value (5 mgN l^{-1} and 0.15 mgP l^{-1}; Mander, 2008). For instance, due to a significant decrease in agricultural intensity in Eastern Europe in the last 12 years, the nutrient losses from fields have dropped but the buffers' outflow values have not changed, i.e., being sometimes higher than inflow concentrations (Kuusemets et al., 2001).

Nitrogen

Three biological processes can remove nitrogen (Mander et al., 2005):

- uptake and storage in vegetation,
- microbial immobilization and storage in the soil as organic nitrogen and
- microbial conversion to gaseous forms of nitrogen (denitrification; nitrification; dissimilative nitrate reduction to ammonia (DNRA)).

Various biophysical conditions control the intensity of these processes; the variability of their intensity is, therefore, very high. A study across watersheds in Japan, India and Indonesia (Anbumozhi et al., 2005) demonstrated the impact of instream processes and riparian buffer zones in higher order streams where the observed NO_3 − levels were 43.7% less than that of the upland.

Studies of perennial streams have identified denitrification as one of the dominant mechanisms removing nitrogen (Woodward et al., 2009), however, this process has intrinsic limitations. In a field study in riparian zones of small Dutch streams, Hefting et al. (2003) found that nitrogen buffering capacity decreased with nitrate load but that the rate of emission of nitrous oxide (N_2O), a potent greenhouse gas, increased dramatically with nitrogen load. High nitrate availability inhibits or retards N_2O reduction and, as a result, substantial quantities of N_2O may be emitted from riparian buffer zones in agricultural environments. Also a recent report on N addition to streams across multiple biomes and land uses in the U.S. (Mulholland et al., 2008) showed that both total uptake velocity and denitrification rate decreased with nitrate concentration.

Clément et al. (2005) measured iron-driven denitrification in riparian wetland, allowing oxidation of ammonia under anaerobic conditions and further denitrification of the nitrate produced by denitrification. If this new pathway is confirmed to be widely occurring, it challenges the currently accepted belief that denitrification in riparian zones is limited by nitrate production under anaerobic conditions or allochthonous input to anoxic hot spots. This would require reconsidering the current conceptual functioning of riparian buffer zones (Pinay & Hannah, 2009).

Phosphorus

Storage of phosphorus in riparian buffer zones depends on the following processes (Mander et al., 2005)

- soil adsorption,
- removal of dissolved inorganic phosphorus by plant uptake,
- microbial immobilisation
- incorporation of organic phosphorus into peat.

Depending on the rooting medium, much of the phosphorous component may become fixed within the soil itself (Brix, 1987). In absolute terms, soil adsorption and vegetation uptake are on a comparable level, varying from 0.1 to 236 and from 0.2 to 50 kg P ha^{-1} yr^{-1}, respectively (Mander & Kimmel, 2008). However, accumulated P can also be released from the wetland soils of riparian zones, especially after lowering of the input concentrations.

A dynamic modelling approach for a moderate-size Belgian river stretch (Ghermandi et al., 2009) showed that in some occasions shading may be more effective than nutrient removal in controlling stream eutrophication and can effectively be implemented as a direct management strategy to improve water quality conditions in small and moderate-size watercourses that are exposed to excessive algal growth during summer periods (**M380**). Similarly Hutchins et al. (2010) found reducing nutrient pollution to be less effective at suppressing phytoplankton growth than the less costly option of establishing riparian shading. In the Swale tributary, ongoing efforts to reduce phosphorus loads in sewage treatment works will only reduce peak (95th percentile) phytoplankton by 11%, whereas a reduction of 44% is possible if riparian tree cover is also implemented.

Humic substances

Like other wetlands, riparian wetlands are important players in the carbon cycle of the watershed. They accumulate large amounts of coarse particulate organic matter (CPOM) and they release dissolved organic matter into the stream and gaseous carbon compounds into the atmosphere (Wantzen & Junk, 2008). The main pathways of organic matter are shown in Fig. 10.

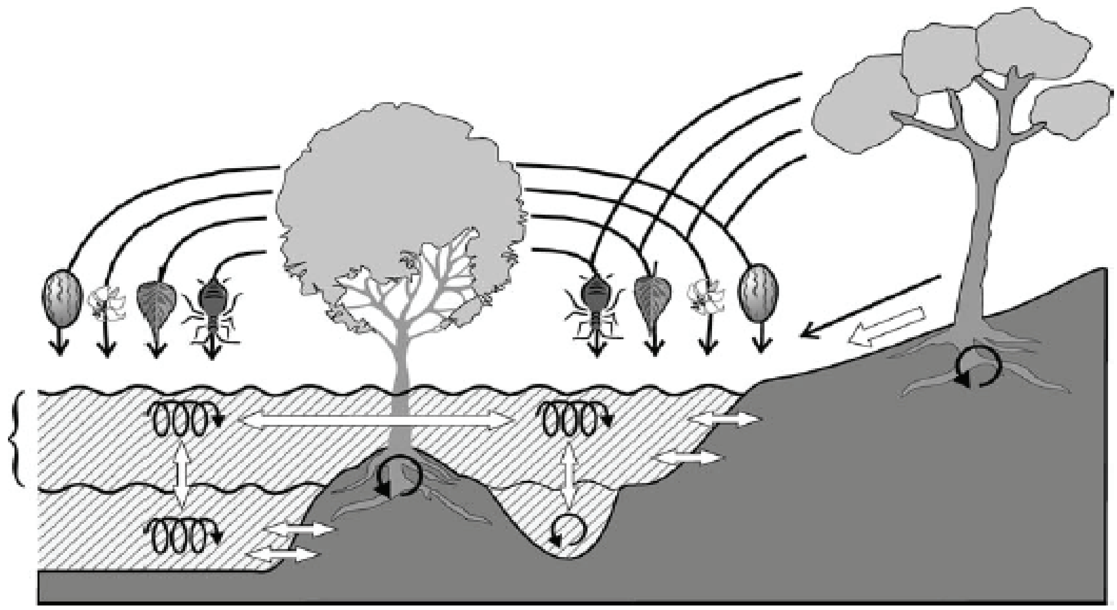


Fig. 10 *Inputs, turnover, and exchange of organic matter in the stream channel (left) and a riparian wetland water body (center) at low and high water levels. Black arrows indicate organic matter inputs, white arrows indicate water exchange pathways, spirals indicate nutrient spiralling or downriver transport, and circular arrows indicate sites of organic matter turnover in situ. Curly brace indicates water-level fluctuations during flood events (Wantzen & Junk, 2008).*

Erosion

Well-established riparian vegetation stabilizes stream banks and contributes to erosion control (Ensign and Mallin, 2001; McKergow et al., 2003; Boothroyd et al., 2004; Ghermandi et al., 2009). Newly created riparian buffers, however, can initially increase erosion (Quinn et al., 1997; Collier et al., 2001). All planting scenarios for degraded New Zealand pastoral streams were predicted to increase sediment yields over a 25-year timeframe, with maximal sediment yield occurring about 15 years after planting due to expected erosion of the streambanks under the developing forest shade (Collier et al., 2001). Sediment yield was greatest for full catchment planting over 25 years, although lowest sediment yield was expected with this scenario over longer timescales. Increased bank erosion can result in widening of the stream channel in stream reaches with newly planted buffers (Sweeney et al., 2004).

Several forestry studies (e.g., Boothroyd et al., 2004; Moore et al., 2005; Chizinski et al., 2010) have shown that retaining even narrow forested buffers along stream banks in logging areas (**M383**) will considerably decrease the loading of suspeded solids into the streams.

Rashin et al. (2006) who examined the effectiveness of timber harvest practices for controlling sediment related water quality impacts in U.S. State Washington, found that 94% of erosion factors associated with sediment delivery to headwater streams were located within 10 m of streams, supporting the value of a near-stream buffer to reduce sedimentation impacts.

It is common that 60–70% of the flow in large river systems originates in headwaters. Streams in the headwater are narrower and associated riparian forest zones are also narrow. Therefore, as Anbumozhi et al. (2005) concluded, less land is required there for nutrient retention and the cumulative benefits of abating non-ponit source pollution may have more significant impact than if larger order streams or the main river are targeted.

In analysing the effect of changes in length and area of riparian forest strips on nutrient concentration in stream waters, Anbumozhi et al. (2005) found a linear relationship between riparian forest area and NO_3^- ion concentration in surface water, particularly higher order streams. However, a similar relationship was not found in the Cl^- ion concentration in lower order streams that the authors explained by a spatial lag between the land use indices and stream water quality, which make direct linear correlations inappropriate.

Filtration by vegetated riparian buffers significantly abates also the concentration of other relevant water contaminants, such as chloride (Anbumozhi et al., 2005), and pesticides (Borin et al., 2004; Lin et al., 2002; Sweeney et al., 2004) and may play an important role in controlling the microbiological quality of the stream water (Ensign & Mallin, 2001; Meals, 2001).

Forestry measures to provide intensive protection for water quality should include (1) maintaining extra wide buffer strips; (2) maintaining plant cover at all times; (3) encouraging rapid establishment of regeneration following treatments; (4) minimizing disturbance, erosion, and sedimentation; (5) restricting use of chemicals; (6) restricting road building or use; and (7) restricting beaver activity (M399, Twery & Hornbeck, 2001).

Cost effectiveness of riparian forest buffer strips has been evaluated in several studies referred by Anbumozhi et al. (2005) where financially optimal buffer widths have been determined. The costs of buffer strips are relatively easy to quantify, but the benefits (non-market values – e.g., fish habitat, species diversity, and water quality) are not. Establishment of forest buffer strips normally results in additional costs to the landowner, public or private. Costs incurred include the loss of stumpage, higher costs of logging and road construction, and additional administrative costs.

Moore et al. (2005) showed the role of legislative tools in modifying stream conditions. Most jurisdictions in the U.S. Pacific Northwest require buffer strips to be left along larger fish bearing streams, but less protection is afforded to smaller, non-fish-bearing streams. Thus, small streams are potentially subject to significant impacts and changes.

Several recent findings reviewed by Pinay & Hannah (2009) suggest that, when addressing the impact of global change on the controls of diffuse pollution, it would be most pertinent to move away from a classical impact assessment of climatic change on riparian zones and adopt a broader spatio-temporal perspective. Riparian zones are often bypassed by, or disconnected from, diffuse nutrient input from upslope (Groffman et al., 2003) that necessitates considering the possible evolution of the landscape as a whole under global change, the riparian zone being one landscape element among others. Adopting a drainage basin approach to understanding the consequences of climate change on water quality would allow the research community to address the problem of intrinsic limitation of nutrient removal in landscape structures, to tackle the impact of land-use change on river flow, and to grasp the consequences of the interdependency of element cycles and the cumulative effect of the long-term human impact on river systems.

5.4 Management of catchment land-use to reduce diffuse nutrient loading and soil erosion

5.4.1 Land use change in EU as revealed from Corine Land Cover inventory

The recently published report by the European Environment Agency „EU 2010 Biodiversity Baseline" (EEA, 2010), referring to data from the last Corine Land Cover inventory, indicate that areas of extensive agriculture, grasslands and wetlands continued to decline across Europe in the period between 1990 and 2006 (Table 1). During the same period, artificial surfaces increased by 12 535 km^2, i.e. + 8%.
The main conversions in land use (Table 2) show a continued expansion of artificial surfaces (urban sprawl and building of economic sites and infrastructures) and abandoned land at the expense of agricultural land, grasslands and wetlands across the EU. Natural grasslands are still being converted into arable land and built-up areas.

5.4.2 Diffuse loading management through landuse changes

Different types of land cover and different land use practices affect runoff formation and the mobility of carbon, nutrients, and suspended solids within the landscape. Knowledge of processes that govern diffuse loading formation allows using them for load abatement through land use modification.

Table 1: Changes in ecosystems between 1990 and 2006 — based on Corine Land Cover

Ecosystem	**Surface change (km^2)**	**Change (%)**
Agro-ecosystems (intensive and heterogeneous, agro-forest)	– 12 611	– 2.0
Agro-ecosystems (extensive)	– 4 476	– 2.6
Grasslands (pastures)	– 2 553	– 0.9
Grasslands (natural)	– 1 795	– 2.4
Heath and scrubs	+ 13 245	+ 5.9
Forests	+ 5 378	+ 0.6
Wetlands (marshes/bogs)	– 1 266	– 5.0

Table 2: Land-use conversion in EU between 2000 and 2006

Change in agro-ecosystems (complex)	20 % mainly due to urban diffuse residential sprawl 31 % due to sprawl of economic sites and infrastructures
Change in agro-ecosystems (extensive)	22 % due to conversion from semi-natural land to agriculture 39 % due to forests creation and afforestation
Change in grasslands	21 % due to sprawl of economic sites and infrastructures 32 % due to arable and permanent crops
Change in forests	94 % due to recent felling and transition and forests internal conversions
Change in heaths and scrubs	84 % due to conversion from transitional woodland to forest
Change in wetlands	35 % due to conversion to agriculture 49 % due to forest creation and afforestation

5.4.2.1 Agricultural landscapes

Nutrient losses. Intensive agriculture, as practiced in many parts of Europe, depends on high rates of use of fertilisers and pesticides. The maintenance of high productivity over time is unlikely to be sustainable in the face of disturbance, disease, soil erosion and overuse of natural resources including water (EEA, 2010). In relation to soil erosion, current agriculture methods accelerate soil loss rates to as high as 4 mm yr^{-1}, which is up to 100 times faster than the rate of soil production (EASAC, 2009). Human use and the status of crops and livestock increased significantly between 1950 and 1990 (Table 3). However, from 1990 to the present, there has been a mixed trend for the status of crops and livestock production across Europe.

Nutrient loading from agriculture remains the major factor causing degradation and loss of biodiversity in surface water ecosystems. Agriculture is the main source of nitrogen loading to water bodies in Western Europe while agriculture and households contribute the most to phosphorus loading. Several publications (e.g., Berka et al., 2001; Zhang et al., 2007; Bouraoui et al., 2009) have demonstrated the role of **rainfall intensity**, **soil N level**, and **fertilizer use** in formation of nitrogen run-off in agricultural landscapes. In some cases, applications of surplus nitrogen reach very high values. In a case study from British Columbia (Berka et al., 2001) surplus nitrogen applications from fertilizers and manure averaged 120 kg ha^{-1} yr^{-1} and exceeded even 300 kg ha^{-1} yr^{-1} in some parts of the watershed as a result of a consistent 59 and 165% increase in pig and chicken numbers, respectively, between 1986 and 1996. Water quality was impacted in two ways: nitrate contaminated groundwater contributed to high nitrates in a major tributary during the summer, while in the wet winter season ammonia, phosphate, and coliform levels were high throughout the drainage system. A significant negative relationship was found between surplus nitrogen applications and dissolved oxygen while ammonia and nitrate concentrations during the wet season were positively correlated to surplus applications.

Soil erosion is considered the main land degradation and desertification process leading to the progressive inability of the vegetation and soils to regenerate, exceeding the resilience status of these ecosystems. Areas of severe soil loss are often the critical areas for agricultural non-point source pollution (Sivertun & Prange, 2010). Erosion includes not only the transport of sediment particles but also the transport of nutrients and pollutants. Both mechanisms depend on the amount of surface runoff and are therefore linked together. Both processes can only be lessened by reducing the surface runoff in favour of ground water infiltration. Dunjo et al. (2004) reported slightly higher significant correlation between runoff and soil loss in winter than in summer studying runoff–erosion microplots in a small Mediterranean catchment. Analysis of variance indicated significant differences in sediment yield between the land use–land cover in winter. Land degradation in the Mediterranean belt through deforestation, intensive cultivation of sloping lands, land misuse and abandonment is largely considered a society-driven problem, which can be effectively managed only through a thorough understanding of the principal ecological, socio-cultural and economic drivingforces associated with land use and climate change, and their impacts.

The importance of vegetation in erosion control is attributed to two main effects: on the one hand, the direct mechanical protection of the soil surface by the canopy and litter covers that intercept rainfall and, consequently, reduce the detachment of soil particles caused by raindrop impact at the soil surface, and on the other hand the indirect improvement of the soil physical and chemical properties, essentially by the incorporation of organic matter. Cultivated or recently abandoned environments proved the highest runoff and soil loss rates while the cork tree and abandoned schrubland environments present the lowest (Dunjo et al., 2004).

There is convincing evidence that even crop selection can alter catchment water discharge. By analysing daily streamflow for the 1890–2003 period from the U.S. Geological Survey stream gage at Keokuk, Iowa and comparing it with agricultural statistics for soybean production in the watershed above the gage, Schilling et al. (2010) demonstrated that increasing soybean acreage increased the slope of discharge–precipitation relationship by 32%. With row crop expansion anticipated in this area from ethanol production, increasing agricultural production is expected to result in increased water yield and nutrient export.

The European initiative EUROHARP was taken to develop harmonised methodologies for quantifying and reporting nutrient losses from diffuse sources. Various quantification tools were tested in 17 catchments. Bouraoui et al. (2009) who analysed the relationship between the catchment characteristics and the nutrient export using the Principal Component Analysis, found that the concentration of phosphorus was positively correlated to the rainfall intensity and the population density (first component), while the nitrogen concentration was positively correlated to the area of agricultural land (second component). Both P and N concentrations were negatively correlated to the area of water bodies within the catchment, indicating that lakes and reservoirs may contribute to the nutrient retention. Similar results were received by Anbumozhi et al. (2005) comparing four watersheds in Indonesia and India. The Indonesian Cisadane watershed with the highest (68%) proportion of agriculture land exported 1.5–4.4 times more nitrate than watersheds with 36–40% of their land in agriculture.

In the EUROHARP study (Bouraoui et al., 2009), phosphorus concentration was explained in major parts by the rain intensity and discharges from agglomerations. The erosion, being associated to rainfall intensity, was considered the main process controlling the phosphorus export, while the runoff and leaching could be responsible for the nitrate movement.

Nutrient retention. Perennial crops reduce the water run-off during storm conditions, thereby reducing the impact of downstream flooding, and are important for carbon sequestration (EEA, 2010). Increasing stocks of carbon in agricultural systems (**M454**) can represent a win-win situation as high levels of soil organic carbon improve nutrient and water use efficiency, reduce nutrient loss and subsequently increase crop production (Trumper et al., 2009).

Li et al. (2009) investigating the effects of different crop rotations on desert soil organic C, N and P, showed a significant increase in soil organic C pool after 10 years of cultivation. Total soil N was increased significantly (by 61–64%) under wheat–maize and wheat–acacia cultivation, but total soil P was reduced by 38% under wheat–alfalfa.

Several climate change mitigation practices in agriculture (see Uri, 2000 for overview) reduce soil erosion, enhance soil carbon and moisture content, and reduce nutrient leakage. The most widely used are crop residue management (CRM) and conservation tillage.

M399	Crop residue management. A year-round conservation system that usually involves a reduction in the number of passes over the field with tillage implements and/or in the intensity of tillage operations, including the elimination of ploughing inversion of the surface layer of soil.

M400	Conservation tillage. Any tillage and planting system that maintains at least 30% of the soil surface covered by residue after planting to reduce soil erosion by water.

Typically high erosion rates for olive plantations (e.g., Dunjo et al., 2003) have been blocked by no tillage and dense undergrowth practices applied in Greece (Schoorl & Veldkamp, 2001).

In conditions where water resources are suffering from multiple stressors, a major relief contributing also to climate change adaptation would be guaranteed by the measure proposed by Hungary[34]

M286	Reduction of non-climate related impacts on hydrological reserves (land use, urbanisation, settlement policy, wastewater)

A large-scale catchment model (LASCAM) was applied to a rural catchment located within the Swan River catchment in Western Australia, to simulate catchment exports of P and N under three management options (Zammit et al., 2005). The model results showed that: (i) full reforestation of agricultural land would reduce P and N export by 50 and 85%, the relative efficiency decreasing with increasing areas of reforestation than for larger areas; (iii) reduction in phosphorus fertiliser application produced a linear response with respect to phosphorus export; (iv) urbanisation would increase P and N loads by about 4 and 12%, respectively, during the 10 years following urbanisation due to the larger impermeable areas causing an increase of overland flow during storms.

5.4.2.2 Urban areas

Increase of impermeable surfaces in urban areas contributes to flash floods. Overflows in combined sewer systems flush large amounts of pollutants into water courses.

Fitzpatrick et al., 2007 analysed associations between biogeochemical analytes in streams in relation to urban and agricultural land uses and described these as „land-use fingerprints‟. Among major ions Na, K, Cl were associated with urban runoff, whereas Ca, Mg with agricultural runoff. Nitrogen signal was equally important in both types of runoff waters but in urban runoff it associated wit phosphorus. Urban run-off was rich in trace elements (V, Cr, Co, Cu, Se, Rb, Mo,

[34] Hungarian 5th National Communication to the UNFCCC. 2009

Sr, Cd, Pb and Ba), which occurred in associated way. In agricultural runoff only U and As co-occurred specifically. It is becoming apparent that despite obscuring factors, land use produces consistent, quantifiable associations between biogeochemical analytes.

Analyses made under the European initiative, EUROHARP (Bouraoui et al., 2009) showed that the concentration of phosphorus at the catchment outlet was positively correlated to the rainfall intensity and the population density conferring the correlation of P with urbanisation. A number of suitable measures have been developed to avoid diffuse pollution from urban areas.

M005 Sediment traps and sustainable urban drainage systems

M192 Increase cleaning efficiency of sewage treatment plants and combined sewers or discharges from storm water channels, because of the lower dilution capacity in receiving waters as a consequence of low flow and increased water temperatures

M203 Implementation of urban constraints, promoting agri-environment (hedges, headlands)[35]

M299 Connection of unsewered wastewater discharges to municipal system in selected areas where assimilative capacity is available during low flow

M342 Treat highly polluting urban discharges

M337 Improve sewer network; increase treatment

M377 Smart Flow sewer to separate the most polluted water of the first rainfall after a dry period.

5.4.2.3 Hedges and shelter belts

Hedges and shelter belts have an important function in reducing soil erosion, and provide regulating services in the form of habitat and shelter for pollinators and sources of natural pest control, whilst increasing ecological connectivity (Vandewalle et al., 2010; Harrison et al., 2010). Buffer strips along water courses have partly similar functions intercepting, at least in part, non-point source pollution from fields. Trees are an important component of the buffer strips not only because of shading but also for their role in nutrient control. Analysis of 44 watersheds in western Oregon ranging in size between 3 and 33 km^2 (Floyd et al., 2009) showed strong negative correlations (r = −0.81 to −0.94) between nitrate-N and the proportion of woody vegetation during winter and spring.

[35] Proposed in Wallonian draft RBMP

5.4.2.4 Grasslands

Grasslands sequester significant amounts of carbon, reduce soil erosion and assist in water management. Carbon sequestration in semi-natural grasslands tends to be modest due to nitrogen and phosphorus limitation (Niklaus & Körner, 2004).

Permanent grasslands also prevent soil erosion and lower the risk that pollutants will leach into water and allows for lower usage of fertiliser, which is one of the main sources of nitrous oxide emissions (Veen et al., 2009). Both vegetation and soil organisms have profound impacts on water movements: vegetation is a major factor in controlling floods, water flows and water quality; vegetation cover in upstream watersheds can affect quantity, quality and variability of water supply; soil micro-organisms are important in water purification; and soil invertebrates influence soil structure, decreasing surface run-off (EASAC, 2009; Turbé et al., 2010).

Intensification of agriculture has resulted in the conversion of some semi-natural grasslands to either cultivated permanent pastures or hayfields and in the abandonment of others (Vandewalle et al., 2010). Emissions of N_20 tend to increase for a number of years following the conversion of temperate grassland to cropland (Conen & Nefyel, 2010).

Table 3: Trends in the status of European ecosystem services between the periods 1950 – 1990 and 1990 - present (EEA, 2010)

Services / Ecosystems	Agro ecosystems	Forests	Grasslands	Heath and scrubs	Wetlands	Lakes and rivers
Provisioning						
Crops/timber	↓ (Mixed)	↑ (Enhanced)			↓ (Degraded)	
Livestock	↓ (Mixed)	= (Degraded)	= (Degraded)	= (Degraded)	↓ (Degraded)	
Wild Foods	= (Degraded)	↓ (Degraded)	↓ (Degraded)	(Unknown)	= (Mixed)	(Unknown)
Wood fuel		= (Mixed)		= (Degraded)		
Capture fisheries					= (Degraded)	= (Degraded)
Aquaculture					↓ (Degraded)	↓ (Degraded)
Genetic	= (Degraded)	↓ (Mixed)	↓ (Degraded)	= (Degraded)	= (Mixed)	(Unknown)
Fresh water		↓ (Degraded)	(Unknown)	(Unknown)	↑ (Mixed)	↑ (Mixed)
Regulating						
Pollination	↑ (Mixed)	↓ (Degraded)	= (Degraded)	(Unknown)		
Climate regulation		↑ (Enhanced)	(Unknown)	= (Mixed)	= (Degraded)	= (Degraded)
Pest regulation	↑ (Mixed)	(Unknown)	= (Degraded)		(Unknown)	
Erosion regulation		= (Mixed)	= (Degraded)	= (Degraded)	(Unknown)	(Unknown)
Water regulation	(Unknown)	= (Mixed)	(Unknown)	↑ (Mixed)	↑ (Enhanced)	= (Degraded)
Water purification		(Unknown)	(Unknown)	(Unknown)	= (Mixed)	= (Degraded)
Hazard regulation		(Unknown)	(Unknown)	(Unknown)	= (Mixed)	= (Degraded)
Cultural						
Recreation	↑ (Mixed)	= (Mixed)	↓ (Degraded)	↑ (Mixed)	↑ (Mixed)	= (Mixed)
Aesthetic	↑ (Mixed)	= (Mixed)	= (Degraded)	= (Degraded)	↑ (Mixed)	= (Degraded)

Status for period 1990–present Degraded Mixed Enhanced Unknown Not applicable

↑ **Positive** ↓ **Negative** = **No trend**

All ecosystem services provided by grasslands show a degraded status since 1990 (Table 3). The number and size of semi-natural grasslands have declined in Europe since the 1950s resulting in a decreased or mixed trend in their human use (EEA, 2010).

5.4.2.5 Heath and scrub ecosystems

Both the shrubs and plant litter have been shown to reduce water run-off and, hence, reduce soil erosion and help curb desertification (Vandewalle et al., 2010). Losses in **heath and scrub** area in Europe have led to a degraded status of many of the services provided by those ecosystems since the 1950s; including livestock production, wood fuel, genetic resources and erosion regulation. However, since the 1990s, there has been a mixed trend in climate regulation, water regulation and recreation services (EEA, 2010).

5.4.2.6 Forests

Forests deliver major services in regulating climate, water cycle, protecting watersheds against nutrient run-off and erosion. Forests, wetlands and protected areas with dedicated management actions often provide clean water at a much lower cost than man-made substitutes like water treatment plants (TEEB, 2009).

Deforestation will almost always negatively affect soil properties, leading in most cases to short-term soil productivity loss (Veldkamp et al., 2001). The conversion of forest to grasslands and permanent crops such as plantations usually leads to less degradation after several years because these systems allow the soil to recover to some extent (Schoorl & Veldkamp, 2001). With increasing deforestation and land use intensity the nitrate output increases, too (Lenhart et al., 2003). Emissions of N_20 tend to increase for a number of years following the conversion of tropical forest to grassland (Conen & Nefyel, 2010). Conversion from forest or grassland to arable lands is the worst scenario in terms of soil productivity and quality. This was confirmed also by a study of Schipper et al. (2007) in New Zealand, in which the conversion of native forests to pasture caused initially little change to soil organic carbon stocks. However, resampling the sites up to 30 years later showed significant losses in carbon and nitrogen, in part caused by soil erosion and leaching.

Forest status has, in general, been enhanced since 1990, which, in combination with reforestation and afforestation across Europe, has resulted in an increase in carbon sequestration. Services such as erosion and water regulation show a mixed trend (EEA, 2010).

5.4.2.7 Wetlands

Wetlands provide protection from floods and storms, control soil erosion and can serve as natural wastewater treatment systems. Large decreases in the surface area of wetlands across Europe between 1950 and 1980 decreased their ability to provide and store fresh water and regulate the climate; the use of fisheries declined as well before 1990. In contrast, agricultural production in wetland ecosystems increased (EEA, 2010).

In organic soils Where N pools are large, N_20 emissions increase dramatically following drainage and cultivation and continue to remain substantially larger than before for a long time. Even abandoning cultivation may not reduce emissions for decades (Conen & Nefyel, 2010).

More recent changes in wetland areas show a regionally mixed trend in their use and the status of their services (Table 3). Water retention has been enhanced in a number of cases through protection and restoration measures (**M030-M032, M037, M038, M115, M132, M133, M220, M295, M307, M329, M335, M375, M432, M434, M449**) and recreation and aesthetic values in wetlands have increased since the 1950s. Newly created riparian buffers, however, can initially increase erosion (Quinn et al., 1997; Collier et al., 2001). All planting scenarios for degraded New Zealand pastoral streams were predicted to increase sediment yields over a 25-year timeframe, with maximal sediment yield occurring about 15 years after planting due to expected erosion of the streambanks under the developing forest shade (Collier et al., 2001).

As mentioned in Hungarian RBMPs, creating new excess water drainage reservoirs (**M022**), in order to increase water availability for irrigation and enhance microclimate is envisaged and accepted only if based on conversion from arable land (Nixon, 2008).

5.4.2.8 Coastal ecosystems

Coastal ecosystems provide food and play an important role as fish nursery habitats; they also provide natural filters for pollution and storage of carbon, a buffer against coastal erosion. Coastal wetlands are known to play a major role in defence against tidal flooding (EEA, 2010).

5.4.2.9 Lake and river ecosystems

Lake and river ecosystems are extremely important for the provision of human drinking water. Rivers and flood plains play an important role within the freshwater cycle (Vandewalle et al., 2010). They provide most global drinking water resources, water resources for agriculture, industry and sanitation, and food such as fish and shellfish; they also provide recreational opportunities and a means of transportation and are a source of energy production (TEEB, 2009). The status of almost all services associated with lake and river ecosystems has been degraded since the 1950s. Demand for flood protection, water regulation, recreation and ecotourism has increased significantly in Europe since the 1950s, but key regulating services such as water purification and flood control continue to be degraded. The use of fresh water from rivers and lakes in Europe has increased since the 1950s. In spite of the trend having slightly reversed since 1990, the total freshwater abstraction is still at a high level in Europe. Regarding freshwater capture fisheries and aquaculture, its use increased from 1950 to 1990 and then decreased slightly (Harrison et al., 2010).

5.5 Management of water abstraction from, and effluent discharge to, surface waters

5.5.1 Abstraction

According to the Sustainable water use report by European Environment Agency (EEA, 1999), about 75% of the total water abstracted for all uses came from surface water, about 25% from groundwater, and only a very minor part from desalination of seawater and from re-use of treated effluents. Abstracted freshwater in Europe is used for urban use (14%), agriculture (30%), and industry (10%, cooling water excluded), and for cooling water for power generation and hydropower (32%), and other or undefined uses (14%). The 4th assessment report of Europe's Environment (EEA, 2007) found that total water abstraction in the region decreased by more than 20% over the period 1990-2005. Total water abstraction in Europe[36] is expected to decrease by more than 10% between 2000 and 2030 with pronounced decreases in Western Europe.

Climate change is expected to reduce water availability and increase irrigation withdrawals in Mediterranean river basins. Under mid-range assumptions on temperature and precipitation changes, water availability is expected to decline in southern and south-eastern Europe (by 10% or more in some river basins by 2030).

The sectoral profile of water abstraction is expected to change: withdrawals for the electricity sector are projected to decrease dramatically over the next 30 years as a result of continuing substitution of once-through cooling by less water-intensive cooling tower systems. Water use in the manufacturing sector may grow significantly. Agricultureis expected to remain the largest water user in the Mediterranen countries, with more irrigation and warmer and drier growing seasons resulting from climate change.

The use of fresh water from rivers and lakes in Europe (EEA, 2010) has increased since the 1950s (Table 3). In spite of the trend having slightly reversed since 1990, the total freshwater abstraction is still at a high level in Europe.

Risks related to overabstraction may include:

- dangerous lowering of groundwater levels, especially in protected areas for groundwater dependent and/or supported habitats (see text box below),
- saline intrusion to coastal aquifers,
- impact to low flow regime in streams,
- lowering of water levels (karstic areas, endorheic lakes)

[36] Use of freshwater resources - outlook from EEA (Outlook 014) - Assessment published Jun 2007

Groundwater dependent and/or supported habitats

Ecosystems that would be significantly altered by a change in the chemistry, volume or timing of groundwater supply are termed groundwater-dependent (phreatic) ecosystems (GDEs). Great progress in GDEs' research has been done in Australia. In 2006 a special edition of Australian Journal of Botany (Vol. 54,) edited by Eamus & Froend was dedicated to these ecosystems.

The major groundwater dependent ecosystem types that have been identified in Australia are as follows:

a) **terrestrial vegetation** - vegetation communities and dependent fauna that have seasonal or episodic dependence on groundwater. Examples include paperbark or ti-tree swamps where the trees access groundwater with their root systems.

b) **river base flow systems** - aquatic and riparian ecosystems that exist in or adjacent to streams that are fed by groundwater base flow during low rainfall periods. Many of Tasmania's rivers and streams are included in this category.

c) **aquifer and cave ecosystems** - aquatic ecosystems that occupy caves, sinkholes, and alluvial and fractured rock aquifers. These ecosystems include, for example, organisms that have specifically adapted to the darkness and constant temperature conditions typically found underground. This group of ecosystems, called also **subsurface groundwater-dependent ecosystems (SGDEs)**, is the least known and has been largely overlooked in favour of more accessible systems. A review of SGDE biodiversity, ecological processes and ecosystem services was published by Tomlinson & Boulton (2008).

d) **wetlands** - aquatic communities and fringing vegetation dependent on groundwater fed lakes and wetlands.

e) **estuarine and near-shore marine ecosystems** - coastal, estuarine and near shore marine plant and animal communities whose ecological function has some dependence on discharge of groundwater.

All the major types of GDEs occur also in other continents. The classification of groundwater bodies in Europe under the WFD includes the requirement to assess the 'significant damage' to **groundwater dependent terrestrial ecosystems (GWDTEs)** caused by anthropogenic pressures.

The Habitats Directive lists in its Annex 1 the natural habitat types whose conservation requires the designation of Special Areas of Conservation (SAC). Among those there are petrifying springs with tufa formation and turloughs, which clearly belong to GDEs.

Petrifying springs with tufa formation (Cratoneurion), NATURA 2000 habitat type code 7220, are hard water springs with active formation of travertine or tufa. These formations are found in such diverse environments as forests or open countryside. They are generally small point or linear formations dominated by bryophytes (*Cratoneurion commutati*)[37].

Turloughs, NATURA 2000 habitat type code 3180, are unique wetland habitats in Ireland - depressions in karst that usually become inundated with groundwater during the winter and drain in summer through swallow holes connected to underground water systems. Typically, these ecosystems contain distinctive aquatic and terrestrial plant and animal communities adapted to fluctuating water levels. The main anthropogenic pressures include artificial drainage to facilitate agriculture, and pollution from nutrient inputs (Sheehy Skeffington et al., 2006).

Other seasonal karstic lakes such as the Slovenian poljes, the Breckland Meres on chalk in England and lakes/depressions in the North American karst regions, are not considered to be identical to turloughs, due to their differing hydrology, seasonality, size or geomorphology (Sheehy Skeffington et al., 2006).

Riparian forests that have a special focus in REFRESH can be structurally, floristically and topographically complex and determining the groundwater dependency of all or some of the components of such forests is a difficult task. By comparing the stable isotope composition of groundwater, soil water and xylem sap, it is sometimes possible to determine the sources of water being transpired (Zencich *et al*. 2002).

Among **wetland types**, Kilroy et al. (2009) considered raised bogs, fens and turloughs representing three most common types of GWDTES in Ireland. In their paper they defined GWDTES as wetlands that depend on a significant proportion of their water supply (quality and quantity) from groundwater.

Groundwater variables that are important to the ecology of the GDEs include duration, timing and rate of seasonal flooding/drying, pressure, flow rate, and depth of groundwater, and/or the specific groundwater quality parameters such as temperature or mineral content. The challenge is how groundwater abstraction (the timing, duration and amount of abstraction) can be managed in order to maintain a desired level of ecosystem function. (i) which attributes of the groundwater regime are important to the GDEs, (ii) what are the safe limits to changes in groundwater regime and (iii) which features of vegetation can be measured to monitor ecosystem function (Eamus & Froend, 2006).

5.5.1.1 General considerations for the management of abstractions

A range of measures can be used to achieve a balance between minimising environmental impacts, maximising scheme yields, sustainability and cost effectiveness. The application of generic operating rules without specifically targeting critical flows and time periods may be ineffective in managing water resources. Unnecessarily stringent operating rules, while apparently honouring

[37] http://eunis.eea.europa.eu/habitats/10150

the precautionary principle, may in fact have indirect adverse effects on other river systems where alternative supply sources have to be used (Hendry et al., 2003). A general consideration applicable to all approaches is that there should be scope for operating rules to vary seasonally to maximise their effectiveness in protecting aquatic ecosystems and to minimise disruption to scheme yields. The efficient use of resources at times when no adverse impact is apparent, particularly in conjunctive use schemes, may offer potential for reduced abstraction under more environmentally sensitive conditions.

In the RBMPs and scientific literature the following measures have been proposed to balance water yield and environmental requirements:

5.5.1.2 Administrative control measures:

- **M007** Registration of abstraction structures.
- **M014** Abstraction licensing.
- **M244** (BG) Public control over water pollution and illegal abstraction and creating "green line" phones to municipalities and regional inspectorates of environment and water.
- **M248** (BG) Control permits for water (control abstraction).
- **M249** Regulation in the permits for water abstraction for dangerous lowering of groundwater levels.
- **M247** Prohibition on issuing permits for water, where the total water use exceeds the operational resources of groundwater bodies.
- **M088** Prioritised water use.
- **M086** Influence water use for the reduction of water needs.

5.5.1.3 Technological measures

- **M254** (BG) Optimization of water abstraction for industrial use through the introduction of closed cycles.
- **M290** Develop a new water resources management (drought tolerant plants, water-saving irrigation technologies and equipment), apply water saving methods to increase the efficiency of water use.
- **M306** Increase the efficiency of water use in the context of reducing river low flows in summer, supported by metering, leakage control and potential rain water harvesting.

5.5.1.4 Hydrological measures

- **M347** Provide improved river flows by reducing net abstraction.
- **M297** (IE), **332** (GB) Altered abstraction timing.
- **M279** Limit unsustainable water uses that are more common in the summer months and those which cannot and should not be considered in dimensioning of water supply infrastructure.
- **M304** Monitor changes in hydrological pressures and review and adjust abstractions and other pressures which reduce groundwater levels in protected areas for groundwater dependent and/or supported habitats and species.
- **M346** Provide improved river flows by changing pattern of abstraction e.g., diurnal and tidal modulation of abstraction (Hendry et al., 2003).
- Prescribed flow (PF) and percentage take rules (Hendry et al., 2003)
    - **M402** Flexible abstraction rules with a higher prescribed flow (PF) at times and in locations where fish are migrating, and a relaxation at other times of year when no migration.
    - **M403** Abstraction operating rules allowing only a proportion, typically 50%, of the flow above the prescribed flow (PF) to be taken.

5.5.1.5 Capacity building measures

- **M321** Development of a support tool for the simulation of scenarios water use in agriculture and alignment of agro-meteorological forecasts with the management of water resources.
- **M441** Develop visualization tools to convey relative risks arising from climate change and other anthropogenic pressures on the freshwater environment (e.g., new urban and infrastructure development, over-abstraction, saline intrusion, diffuse runoff, uncontrolled waste water discharges, and habitat degradation).

5.5.2 Effluent release

The Urban Waste Water Treatment Directive (91/271/EEC) is an important Community water policy and its aim is to protect the environment from the adverse effects of urban waste water discharges. The directive sets minimum standards for the collection, treatment and disposal of waste water dependent upon the size of the agglomeration, and the type and sensitivity of the receiving waters.

According to the latest report on urban waste water treatment[38]

- Wastewater treatment in all parts of Europe has improved significantly since the 1980s.
- In several countries in north-western Europe there has been a marked increase in the population connected to tertiary waste water treatment in the 1990s resulting in marked reductions in phosphorus and nitrogen discharges.
- However the percentage of population connected to waste water treatment is relatively low in southern Europe and in the Accession countries.

With onset of climate change, the decline of the dilution capacity due to low water in streams should be taken into account in waste water discharge into natural recipients. Higher water temperatures cause lower oxygen solubility and may lead to oxygen depletion, decreasing the capacity of a stream to assimilate organic wastes.

In the RBMPs and scientific literature the following measures have been proposed to reduce the adverse effects of effluent discharges to the environment:

5.5.2.1 General measures

- **M286** Reduction of non-climate related impacts on hydrological reserves (land use, urbanisation, settlement policy, wastewater).
- **M135** Adaptation measures at sewage collection and treatment sites.

5.5.2.2 Administrative measures

- **M102** Reduce phosphates entering wastewater (e.g. through phasing out of phosphate detergents).
- **M109** Emissions control from waste water treatment plants.
- **M110** Measures to deal with nutrient pressures, e.g. licensing discharges from waste water treatment.

5.5.2.3 Technological and construction measures

- **M103** Tune wastewater discharges to the carrying capacity of the aquatic system.

[38] EEA. Urban waste water treatment. Assessment made on 01 May 2004

- **M105** Building of waste water treatment plants for urban wastewaters.
- **M106** Nutrient removal from sewage water.
- **M111** Improvement of waste water purification technologies.
- **M114** Design of new or enhanced wastewater treatment works.
- **M142** Construction and refurbishment of wastewater treatment plants and sewerage systems in conurbations with more than 2,000 PE .
- **M143** Construction and refurbishment of waste water treatment plants and sewerage systems in conurbations with up to 2,000 PE in areas requiring special protection.
- **M192** Increase cleaning efficiency of sewage treatment plants and combined sewers or discharges from storm water channels, because of the lower dilution capacity in receiving waters as a consequence of low flow and increased water temperatures.
- **M280** Planning new development areas with stormwater and sewage water separation systems and remote retention reservoirs.
- **M293** The decline of the dilution capacity due to low water in streams should be taken into account in waste water discharge into natural recipients.
- **M303** Make provision for pre-treatment requirements for industrial wastewater entering the collection systems and treatment plants considering the potentially reduced assimilative capacity in rivers in summer.
- **M336** Reduce pollutant content of sewage at source.
- **M337** Improve sewer network; increase treatment.
- **M377** Smart Flow sewer to separate the most polluted water of the first rainfall after a dry period.

5.5.2.4 Hydrological measures

- **M292** Treated waste water should be kept on site.
- **M299** Connection of unsewered wastewater discharges to municipal system in selected areas where assimilative capacity is available during low flow.

5.5.3 Thermal effluent discharge

In the 4th assessment report of Europe's Environment (EEA, 2007) the European Environment Agency admitted that despite the introduction of more efficient cooling technologies only a minor reduction in the use of water in energy generation was achieved between 1990 and 2002, with many of the WCE countries still using more than half of their abstracted water in power plants. In the context of climate change, the thermal effluent discharge represents a strong additional pressure to the receiving water bodies and can cause considerable changes in temperature regimes. For example, two-thirds of 3 °C temperature increase of the River Rhine between 1910 and 2006 is estimated to be due to the increased use of cooling water in Germany and only one-third to the increase in temperature as a result of climate change (EEA, 2008).

Thermal disturbance of streams can cause severe and long-lasting alterations in biological communities. Fletcher et al. (2000) described a case in west-central South Carolina, USA, where thermal effluent release to two tributary streams produced stream water temperatures of over 50°C and stream flows of ten times above their base level. After 7–13 years of ambient flows, stream habitats remained severely altered compared to two similar, undisturbed streams due to destruction of the riparian vegetation by past thermal effluents. As a result, the total aquatic macrophyte abundance, which was negatively related to canopy cover, was much higher in these disturbed streams.

Although thermal disturbance remains an important environmental stressor and there is convincing evidence on its impacts on aquatic ecosystems (e.g., Rossi & Hari, 2007; Dallas, 2008; McCullough et al., 2009; Tunowski, 2009) and even temperature criteria have been developed to protect native coldwater and warmwater fishes from thermal stress (Todd et al., 2008), no adaptation measures were proposed by Member States in the RBMPs in this regard.

6 References:

Aguilar-Ibarra, A., P. Lim & S. Lek, 2005. Fish diversity conservation and river restoration in southwest France: a review. In: Lek, S., M. Scardi, P. Verdonschot, J. Descy & Y.S. Park (eds.). Modelling community structure in freshwater ecosystems. Springer, Berlin: 317–42.

Alekseeva, I., J. Jarsjö, C. Schrum & G. Destouni, 2009. Reproducing the Aral Sea water budget and sea–groundwater dynamics between 1979 and 1993 using a coupled 3-D sea-ice–groundwater model. Journal of Marine Systems 76: 296–309.

Allam, A.R., E.-J. Saaf & M.A. Dawoud, 2003. Desalination of brackish groundwater in Egypt. Desalination 152: 19-26.

Altunkaynak, A., M. Özger & Z. Sen, 2003. Triple diagram model of level fluctuations in Lake Van Turkey. Hydrology and Earth System Sciences 7: 235–244.

Amatya, D., Skaggs, R., Gilliam, J., Hughes, J., 2003. Effects of orifice-weir outlet on hydrology and water quality of a drained forested watershed. Southern Journal of Applied Forestry 27: 130–142.

Anbumozhi, V., J. Radhakrishnan & E. Yamaji, 2005. Impact of riparian buffer zones on water quality and associated management considerations. Ecological Engineering 24: 517–523.

Anderson, P.D., D.J.Larson & S.S. Chan, 2007. Riparian buffer and densitymanagement influences on microclimate of young headwater forests of western Oregon. Forestry Science 53: 254–269.

Andersson, E. & S. Sobek, 2006. Comparison of a Mass Balance and an Ecosystem Model Approach when Evaluating the Carbon Cycling in a Lake Ecosystem. Ambio 35: 476-483.

Armstrong, A., 2000. Ethical Considerations in Wetland Management. Physics and Chemistry of the Earth (B) 25: 641-644.

Åström, M., P. Österholm, I. Bärlund & S. Tattari, 2007. Hydrochemical Effects of Surface Liming, Controlled Drainage and Lime-Filter Drainage on Boreal Acid Sulfate Soils. Water Air Soil Pollution 179:107–116.

Azami, K., H. Suzuki & S. Toki, 2004. Changes in riparian vegetation communities below a large dam in a monsoonal region: Futase Dam, Japan. River Research and Applications 20: 549-563.

AT, 2009. The Austrian National Water Management Plan 2009

Ayars, J.E., E.W. Christen & J.W. Hornbuckle, 2006. Controlled drainage for improved water management in arid regions irrigated agriculture. Agricultural Water Management 86: 128-139.

Baginska, B., T. Pritchard & M. Krogh, 2003. Roles of land use resolution and unit-area load rates in assessment of diffuse nutrient emissions. Journal of Environmental Management 69: 39-46.

Barnes, 2009. Hydropower in Europe: Current Status, Future Opportunities. http://www.renewableenergyworld.com/

Barredo, J.I., 2007. Major flood disasters in Europe: 1950–2005. Natural Hazards 42: 125–148.

Bayard, D., M. Stähli, A. Parriaux & H. Flühler, 2005. The influence of seasonally frozen soil on the snowmelt runoff at two Alpine sites in southern Switzerland. Journal of Hydrology 309: 66-84.
BE, 2008a. The Flemish Draft RBMPs for Meuse and Shelde
BE, 2008b. Wallonian draft RBMP
Benson, C. & E.J. Clay, 2004. Understanding the Economic and Financial Impacts of Natural Disasters. Disaster Risk Management Series No. 4, World Bank, Washington, DC.
Berka, C., H. Schreier & K. Hall, 2001. Linking water quality with agricultural intensification in a Rural Watershed. Water Air and Soil Pollution 127: 389–401.
Bernhardt, E.S., M.A. Palmer, J.D.Allan, G.Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S.Katz, G.M.Kondolf, P. S. Lake, R. Lave, J. L.Meyer, T.K. O'Donnell, L. Pagano, B. Powell & E. Sudduth, 2005. Synthesizing U.S. River Restoration Efforts. Science 308: 636-637.
Beschta, R.L., R.E. Bilby, J.W. Brown, L.B. Holtby & T.D.Hofstra, 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. In: Salo, E.O., Cundy, T.W. (Eds.), Streamside Management: Forestry and Fishery Interactions. University of Washington, Institute of Forest Resources, Seattle, WA, pp. 191–232.
BG, 2009a. RBMP of the East Aegian Sea RBD
BG, 2009b. RBMP of the Black Sea RBD
BG, 2009c. RBMP of the Danube RBD
Bhattacharjee, J., J.P. Taylor Jr., L.M. Smith & D.A. Haukos, 2009. Seedling competition between native cottonwood and exotic saltcedar: implications for restoration. Biological Invasions 11:1777–1787.
Bissels, S., N. Hölzel, T.W. Donath & A. Otte, 2004. Evaluation of restoration success in alluvial grasslands under contrasting flooding regimes. Biological Conservation 118: 641-650.
Blöschl, G., S. Ardoin-Bardin, M. Bonell, M. Dorninger, D. Goodrich, D. Gutknecht, D. Matamoros, B. Merz, P. Shand & J. Szolgay, 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes 21: 1241-1247.
Bódis, K., 2007. High-Resolution DEM for Design of Flood Emergency Reservoirs. In: R.J. Peckham & G. Jordan, Digital Terrain Modelling Development and Applications in a Policy Support Environment. 203-226.
Boothroyd, I.K.G, J.M. Quinn, E. R. (Lisa) Langer, K.J. Costley, G. Steward, 2004. Riparian buffers mitigate effects of pine plantation logging on New Zealand streams: 1. Riparian vegetation structure, stream geomorphology and periphyton. Forest Ecology and Management 194: 193-213.
Borin, M., E. Bigon, G. Zanin & L. Fava, 2004. Performance of a narrow buffer strip in abating agricultural pollutants in the shallow subsurface water flux. Environmental Pollution 131: 313–321.
Boulton A.J. & P.J. Hancock, 2006. Rivers as groundwater-dependent ecosystems: a review of degrees of dependency, riverine processes and management implications. Australian Journal of Botany 54: 133–144.
Bouraoui, F., B. Grizzetti, G. Adelsköld, H. Behrendt, I. de Miguel, M. Silgram, S. Gómez, K. Granlund, L. Hoffmann, B. Kronvang, S. Kvaernø, A. Lázár, M. Mimikou, G. Passarella, P. Panagos, H. Reisser, B. Schwarzl, C. Siderius,

A.S. Sileika, A.A. Smit, R. Sugrue, M. Vanliedekerke, J. Zaloudik, 2009. Basin characteristics and nutrient losses: the EUROHARP catchment network perspective. Journal of Environmental Monitoring 11:515-25.

Breukers, C.M., 2000. Creating water management strategies for the northern part of Holland using a collaborative planning process. IAHS-AISH Publication 272: 51–56.

Brookes, A., 1984. Recommendations bearing on the sinuosity of Danish stream channels. – Technical Report No. 6, Freshwater Laboratory, Danish Environmental Protection Agency. 130 pp.

Burt, T.P., 1995. The role of wetlands in runoff generation from headwater catchments. In: Hughes, J. & L.Heathwaite (eds.), Hydrology and Hydrochemistry of British Wetlands. Wiley, Chichester: 21–38.

Burton Jr., G.A., Pitt, R., 2001. Stormwater Effects Handbook: ATool Box for Watershed Managers, Scientists and Engineers. CRC/Lewis Publishers, Boca Raton, Florida.

Butturini, A., S. Bernal, S. Sabater & F. Sabater, 2002. The influence of riparian-hyporheic zone on the hydrological responses in an intermittent stream. Hydrology and Earth System Sciences 6: 515–525.

Börjesson P. & G. Berndes, 2006. The prospects for willow plantations for wastewater treatment in Sweden. Biomass Bioenergy 30: 428–38.

Caissie, D., N. El-Jabi & M.G. Satish, 2001. Modelling of maximum daily water temperatures in a small stream using air temperatures. Journal of Hydrology 251: 14-28.

Cattivelli, L., F. Rizza, F.-W. Badeck, E. Mazzucotelli, A.M. Mastrangelo, E. Francia, C. Marè, A. Tondelli & A.M. Stanca, 2008. Drought tolerance improvement in crop plants: An integrated view from breeding to genomics. Field Crops Research 105: 1-14.

Cheng, J.D., L.L. Lin & H.S. Lu, 2002. Influences of forests on water flows from headwater watersheds in Taiwan. Forest Ecology and Management 165: 11-28.

CIRIA, 2000. Sustainable urban drainage systems. In: Design manual for Scotland and Northern Ireland. Construction Industry Research and Information Association, London, England.

CIS, 2009. River basin management in a changing climate. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance document No. 24. Luxembourg.

Clément, J.C., J. Shrestha, J.G. Ehrenfeld & P.R. Jaffé, 2005. Ammonium oxidation coupled to dissimilatory reduction of iron under anaerobic conditions in wetland soils. Soil Biology and Biochemistry 37: 2323-2328.

Cole, J.J., Y.T. Prairie, N.F. Caraco, W.H. McDowell, L.J. Tranvik, R.G. Striegl, C.M. Duarte, P. Kortelainen, J.A. Downing, J.J. Middelburg & J. Melack, 2007. Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. Ecosystems 2007, 10:171-84.

Collier, K.J., J.C. Rutherford, J.M. Quinn & R.J. Davies-Colley, 2001. Forecasting rehabilitation outcomes for degraded New Zealand pastoral streams. Water Science and Technology 43: 175–184.

COM, 1995. Wise Use and Conservation of Wetlands. Communication from the Commission to the Council and the European Parliament. COM (95) 189 final, 29.05.1995.

Conen, F. & A. Neftel, 2010. Nitrous oxide emissions from land-use and land-management change. In: K. Smith (ed.), Nitrous Oxide and Climate Change, Ch. 7, Earthscan, Washington DC: 143-159.

ConnDOT, 2001. Drainage Manual. Connecticut Department of Transportation. http://www.ct.gov/dot/lib/dot/documents/ddrainage/1.1.pdf

Conti, S., P. Meli, G. Minelli, R. Solimini, V. Toccaceli, M. Vichi, C. Beltrano & L. Perini, 2005. Epidemiologic study of mortality during the Summer 2003 heat wave in Italy. Environmental Research 98: 390-399.

Crawford, J.A. & R.D. Semlitsch, 2007. Estimation of core terrestrial habitat for stream-breeding salamanders and delineation of riparian buffers for protection of biodiversity. Conservation Biology 21: 152–158.

Crétaux, J.-F., A.V. Kouraev, F. Papa, M. Bergé-Nguyen, A. Cazenave, N. Aladin & I.S. Plotnikov, 2005. Evolution of Sea Level of the Big Aral Sea from Satellite Altimetry and Its Implications for Water Balance. Journal of Great Lakes Research 31: 520-534.

CZ, 2007. The Plan of Main River Basins of the Czech Republic Approved by Resolution No 562 of the Government of the Czech Republic of 23 May 2007

Dallas, H. 2008. Water temperature and riverine ecosystems: An overview of knowledge and approaches for assessing biotic responses, with special reference to South Africa. Water SA 34: 393-404.

Davies-Colley, R.J. & J.C. Rutherford, 2005. Some approaches for measuring and modelling riparian shade. Ecological Engineering 24:525-530.

de Vries, S.C., G.W.J. van de Ven, M.K. van Ittersum & K.E. Giller, 2010. Resource use efficiency and environmental performance of nine major biofuel crops, processed by first-generation conversion techniques. Biomass and Bioenergy 34:588-601.

Deressa, T.T., R.M. Hassan, C. Ringler, T. Alemu & M. Yesuf, 2009. Determinants of farmers’ choice of adaptation methods to climate change in the Nile Basin of Ethiopia. Global Environmental Change 19: 248–255.

DFID, 2004. Adaptation to climate change: The right information can help the poor to cope. Department for International Development (DFID). Key sheet 07. Crown Copyright 2004. www.asocam.org/biblioteca/CC_2067.pdf

Directive 2007/60/EC of the European Parliament and of the Council of 23 October 2007 on the assessment and management of flood risks. Official Journal of the European Union L 288/27-34.

Dobkin, D.S., A.C. Rich & W.H. Pyle, 1998. Habitat and avifaunal recovery from livestock grazing in a riparian meadow system of the northwestern great basin. Conservation Biology 12: 209-221.

Dodds, W.K. & R.M. Oakes, 2006. Controls on nutrients across a prairie stream watershed: land use and riparian cover effects. Environmental Management 37: 634–646.

Doty, C.W., K.R. Cain & L.J. Farmer, 1986. Design, operation, and maintenance of controlled-drainage/subirrigation (CD-SI) systems in humid areas. Applied Engineering and Agriculture 2: 114-119.

Downing, J. A., J. J. Cole, J. J. Middelburg, R. G. Striegl, C. M. Duarte, P. Kortelainen, Y. T. Prairie & K. A. Laube, 2008. Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century. Global Biogeochemical Cycles 22, GB1018, doi:10.1029/2006GB002854.

Drainage Management in the San Joaquin Valley. 1998. A status report. San Joaquin Valley Drainage Implementation Program. California.
Ducrotoy, J.-P. & J.-C. Dauvin, 2008. Estuarine conservation and restoration: The Somme and the Seine case studies (English Channel, France). Marine Pollution Bulletin 57: 208-218.
Dunjo, G., G. Pardini & M. Gispert, 2004. The role of land use–land cover on runoff generation and sediment yield at a microplot scale, in a small Mediterranean catchment. Journal of Arid Environments 57: 99–116.
E.U. Commission, 2004. Best practices on flood prevention, protection and mitigation, 29 p. (www.floods.org/PDF/Intl_BestPractices_EU_2004.pdf).
EA, 2010. Working with Natural Processes to Manage Flood and Coastal Erosion Risk. A Guidance Document. Bristol, 73 pp.
Eamus, D. & R. Froend, 2006. Groundwater-dependent ecosystems: the where, what and why of GDEs. Australian Journal of Botany 54: 91–96.
EASAC, 2009, Ecosystem services and biodiversity in Europe, EASAC policy report 09, the Royal Society.
EC, 1992. Council directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal L0043.
EC, 2007. LIFE and Europe's wetlands. Restoring a vital ecosystem. European Communities.
EEA, 1999. Sustainable water use in Europe. Part 1: Sectoral use of water. European Environment Agency, Environmental assessment report No. 1
EEA, 2007. Europe's environment. The fourth assessment. European Environment Agency, Copenhagen.
EEA, 2008. Impacts of Europe's changing climate — 2008 indicator-based assessment
EEA, 2009. Report on good practice measures for climate change adaptation in river basin management plans. EEA final version: 3.0, June 2009. ETC/Water Task: 1.5.4 Water and climate change. Prepared by: Cornelius Laaser, Anna Leipprand, Colette de Roo, Rodrigo Vidaurre (Ecologic Institute), EEA Project Manager Peter Kristensen. EEA/ADS/06/001 - Water
EEA, 2010. EU 2010 Biodiversity Baseline. EEA Technical report No 12/2010. EEA, Copenhagen, 2010: 164 pp.
Eisenreich, S.J. (Ed.), 2005. Climate Change and the European Water Dimension. EU-Report 21553 of the. European Commission, Joint Research Centre, Ispra.
Elliott, A.H. & S.A. Trowsdale, 2007. A review of models for low impact urban stormwater drainage. Environmental Modelling and Software 22: 394–405.
Ensign, S.H. & M.A. Mallin, 2001. Stream water quality changes following timber harvest in a coastal plain swamp forest. Water Resources 35: 3381–3390.
ESB, 2008. Draft River Basin Management Plans. Adapting the Plans to Climate Change. ESB International. Final Report. Dec. 2008
Falkenmark, M. & G. Lindh, 1976. Water for a starving world. Westview Press, Boulder, Colorado, 204 p.
Falloon, P. & R. Betts, 2010. Climate impacts on European agriculture and water management in the context of adaptation and mitigation—The importance of an integrated approach. Science of The Total Environment 408: 5667-5687.
Ficetola, G.F., Thuiller, W. & Padoa-Schioppa, E. (2009) From introduction to the establishment of alien species: bioclimatic differences between presence

and reproduction localities in the slider turtle. Diversity and Distributions, 15, 108-116.
Fitzpatrick R.W., P.J. Davies, B.P. Thomas, R.H. Merry, D.G. Fotheringham, W.S. Hicks & E. Barnett, 2002. Properties, distribution and environmental hazards of South Australian coastal acid sulfate soils. 5th International Acid Sulfate Soils Conference, Tweed Heads, NSW, 25th to 30th August 2002. Book of extended abstracts.
Fitzpatrick, M.L., D.T. Long & B.C. Pijanowski, 2007. Exploring the effects of urban and agricultural land use on surface water chemistry, across a regional watershed, using multivariate statistics. Applied Geochemistry 22: 1825–1840.
Flannigan, M.D., B.J. Stocks & B.M. Wotton, 2000. Climate change and forest fires. Science of The Total Environment 262: 221-229.
Fletcher, D.E., S.D. Wilkins, J.V. McArthur & G.K. Meffe, 2000. Influence of riparian alteration on canopy coverage and macrophyte abundance in Southeastern USA blackwater streams. Ecological Engineering 15 S67–S78.
Floyd, W. C., S. H. Schoenholtz, S. M.Griffith, P. J. Jr. Wigington & J. J. Steiner, 2009. Nitrate-Nitrogen, Land Use/Land Cover, and Soil Drainage Associations at Multiple Spatial Scales. Journal of environmental quality 38: 1473-1482.
Franklin, A.B., B.R. Noon & T.L. George, 2002. What is habitat fragmentation? Studies in Avian Biology 25: 20-29.
GB, 2009a. The river basin management plan for the Scotland river basin district 2009–2015. Chapter 3: Achieving our environmental objectives
GB, 2009b. River Basin Management Plan. Western Wales River Basin District. Annex H: Adapting to climate change
Geilen, N., H. Jochems, L. Krebs, S. Muller, B. Pedroll, T. Van der Sluis, K. Van Looy & S. Van Rooij, 2004. Integration of ecological aspects in flood protection strategies: defining an ecological minimum. River Research and Applications 20: 269–283.
Ghermandi, A., V. Vandenberghe, L. Benedetti, W. Bauwens & P.A. Vanrolleghem, 2009. Model-based assessment of shading effect by riparian vegetation on river water quality. Ecological Engineering 35: 92-104.
Gibbons J. M. & S. J. Ramsden, 2008. Integrated modelling of farm adaptation to climate change in East Anglia, UK: Scaling and farmer decision making. Agriculture, Ecosystems and Environment 127: 126–134.
Gómez, F. & S. Souissi, 2008. The impact of the 2003 summer heat wave and the 2005 late cold wave on the phytoplankton in the north-eastern English Channel. Comptes Rendus Biologies 331: 678-685.
Goodland, R., 1995. The Concept of Environmental Sustainability. Annual Review of Ecology and Systematics 26: 1-24.
Goodland, R., 1997. Environmental sustainability in agriculture: diet matters. Ecological Economics 23: 198-200.
Goodland, R., H. Daly and J. Kellenberg, 1994. Burden sharing in the transition to environmental sustainability. Futures 26: 146-155.
Goodwin, C.N., C.P. Hawkins & J.L. Kershner, 1997. Riparian restoration in the Western United States: Overview and perspective. Restoration Ecology 5 (4 SUPPL.): 4-14.

Goulding, H.L., T.D. Prowse, B. Bonsal, 2009. Hydroclimatic controls on the occurrence of break-up and ice-jam flooding in the Mackenzie Delta, NWT, Canada. Journal of Hydrology 379: 251-267.
Groffman, P.M., D.J. Bain, L.E. Band, K.T. Belt, G.S. Brush, J.M. Grove, R.V. Pouyat, I.C. Yesilonis & W.C. Zipperer, 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment 6:315-321.
Groisman, P.Y., Knight, R.W., Easterling, D.R., Karl, T.R., Hegerl, G.C., Razuvaev, V.N., 2005. Trends in intense precipitation in the climate record. Journal of Climate 18: 1326–1350.
Groisman, P.Y., Knight, R.W., Karl, T.R., Easterling, D.R., Sun, B., Lawrimore, J., 2004. Contemporary changes of the hydrological cycle over the contiguous United States: trends. Journal of Hydrometeorology 5: 64–85.
GWP, 2000. Integrated Water Resources Management, Global Water Partnership Technical Advisory Committee Background Papers, No. 4.
Habron, G., 2003. Role of Adaptive Management for Watershed Councils. Environmental Management 31:29-41.
Hagan, J.M., S. Pealer & A.A. Whitman, 2006. Do small headwater streams have a riparian zone defined by plant communities. Canadian Journal of Forest Research 36: 2131–2140.
Hamilton, G. 2009. Consumerism, self-creation and prospects for a new ecological consciousness. Journal of Cleaner Production 18: 571–575.
Hancock, P., 2002. Human impacts on the stream–groundwater exchange zone. Environmental Management 29: 761–781.
Hancock, P.J., A.J. Boulton & W.F. Humphreys, 2005. Aquifers and hyporheic zones: towards an ecological understanding of groundwater. Hydrogeology Journal 13: 98–111.
Handmer, J. 2001. Improving flood warnings in Europe: a research and policy agenda. Environmental Hazards 3: 19–28.
Hansen, H.O. (ed.), 1996. River Restoration – Danish experience and examples. Ministry of Environment and Energy, National Environmental Research Institute.
Hanson, P. C., D. L. Bade, S. R. Carpenter & T. K. Kratz, 2003. Lake metabolism: relationships with dissolved organic carbon and phosphorus. Limnology and Oceanography, 48: 1112–1119.
Hansson, K., M. Danielson & L. Ekenberg, 2008. A framework for evaluation of flood management strategies. Journal of Environmental Management 86: 465-480.
Harrison, P.A., M. Vandewalle, M.T. Sykes, P.M. Berry, R. Bugter, F. de Bello, C.K. Feld, U. Grandin, R. Harrington, J.R. Haslett, R.H.G. Jongman, G.W. Luck, P. Martins da Silva, M. Moora, J. Settele, J.P. Sousa & M. Zobel, 2010. Identifying and prioritising services in European terrestrial and freshwater ecosystems, Springer Science+Business Media B.V.
Hastings, I.M. & W.M. Watkins, 2005. Intensity of malaria transmission and the evolution of drug resistance. Acta Tropica 94: 218-229.
Hefting, M.M., J.C. Clement, P. Bienkowski, D. Dowrick, C. Guenat, A. Butturini, S. Topa, G. Pinay & J.T.A Verhoeven, 2005. The role of vegetation and litter in the nitrogen dynamics of riparian buffer zones in Europe. Ecological Engineering 24: 465–482.

Hefting, M.M., R Bobbink & H de Caluwe, 2003. Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. Journal of Environmental Quality 32: 194-203.

Helmreich, B. & H. Horn, 2009. Opportunities in rainwater harvesting. Desalination 248:118-124.

Hemphill, R.W. & M.E. Bramley, 1989. Protection of river and canal banks. CIRIA/Butterworths, London.

Hendry, K., D. Cragg-Hine, M. O'Grady, H. Sambrook & A. Stephen, 2003. Management of habitat for rehabilitation and enhancement of salmonid stocks. Fisheries Research 62: 171–192.

Henrikson, L., 2009: Freshwater Pearl Mussel *Margaritifera margaritifera* projects in Europe. WWF Sweden, 30 pp.

Hering, D., A. Schmidt-Kloiber, J. Murphy, S. Lucke, C. Zamora-Munoz, M.J. Lopez-Rodriguez, T. Huber & W. Graf, 2009. Potential impact of climate change on aquatic insects: A sensitivity analysis for European caddisflies (Trichoptera) based on distribution patterns and ecological preferences. Aquatic Sciences 71: 3-14.

Hese, S., W. Lucht, C. Schmullius, M. Barnsley, R. Dubayah, D. Knorr, K. Neumann, T. Riedel, K. Schröter, 2005. Global biomass mapping for an improved understanding of theCO2 balance—the Earth observation mission Carbon-3D. Remote Sensing of Environment 94: 94–104.

Hesterberg, D. 1998. Biogeochemical cycles and processes leading to changes in mobility of chemicals in soils. Agriculture, Ecosystems & Environment 67:121-133.

Hickling R., Roy D. B., Hill J. K., & Thomas C. D.2005: A northward shift of range margins in British Odonata. Global Change Biology 11: 502–506.

Hickling, R., D.B. Roy, J.K. Hill, R. Fox & C.D. Thomas, 2006. The Distributions of a Wide Range of Taxonomic Groups are Expanding Polewards. Global Change Biology 12: 450-455.

Holland, H.D. & K.K. Turekian, 2007. Treatise on Geochemistry. Volume 9: Environmental Geochemistry. Elsevier.

Houghton, R. A., 2007. Balancing the global carbon budget. Annu. Rev. Earth Planet. Sci. 35, 313–347.

House, J. I., Prentice, I. C., Ramankutty, N., Houghton, R. A., & Heimann, M., 2003. Reconciling apparent inconsistencies in estimates of terrestrial CO_2 sources and sinks. Tellus, 55B, 345– 363.

Hovhanissian, R. & B. Gabrielyan, 2000 Ecological problems associated with the biological resource use of Lake Sevan, Armenia. Ecological Engineering 16: 175–180.

HU, 2009a. Hungarian 5th National Communication to the UNFCCC. 2009

HU, 2009b. Magyaroszag vizgyöjtö-gazdalkodasi terve. A 2009. december 22-én közzétett

Huntington, T.G., 2006. Evidence for intensification of the global water cycle: review and synthesis. Journal of Hydrology 319:83-95.

ICBR, 2009. Internationally Coordinated Management Plan for the International River Basin District of the Rhine December 2009. International Commission for the Protection of the Rhine (ICBR)

ICPDR, 2008. Analysis of the Tisza River Basin 2007. Initial step toward the Tisza River Basin Management Plan – 2009. International Commission for the Protection of the Danube River, Vienna.

ICPDR, 2009. Sub-Basin Level Flood Action Plan. Tisza River Basin.
IKSE, 2009. Elbe-Hochwasser wie 2002 nicht mehr möglich. Internationale Kommission zum Schutz der Elbe, Europäischer Wirtscchaftsdients Wasser und Abwasser Nr. 34.
IKSR, 2009. Internationally Coordinated Management Plan for the International River Basin District of the Rhine. December 2009
IPCC, 2007. IPCC Fourth Assessment Report - Climate Change 2007: The Physical Science Basis Summary for Policymakers.
IT, 2010a. Autorita di Bacino del Fiume Po. Piano di Gestione del Distretto Idrographico del Fiume Po. Elenco delle misure specifiche. Versione del 24 Febbraio 2010 Allegato 7.9 all'Elaborato 7.
IT, 2010b. Autorita di Bacino del Fiume Po. Piano di Gestione del Distretto Idrographico del Fiume Po. Elenco delle misure di riferimento per gli scenari e i temi chiave del Piano Versione. Allegato 7.10 all'Elaborato 7. Versione del 24 Febbraio 2010.
IT, 2010c. Piano di gestione acque (Direttiva Comunitaria 2000/60/CE, D.L.vo 152/06, L. 13/09, D.L. 194/09). Relazione Generale. Allegato 17. Misure supplementari/indirizzi, Distretto Idrografico dell'Appennino Meridionale. Autorità di Bacino Nazionale dei Fiumi Liri-Garigliano e Volturno, Regione Abruzzo, Regione Basilicata, Regione Calabria, Regione Campania, Regione Lazio, Regione Molise, Regione Puglia Febbraio 2010.
IWA, 2008. Industry Sector Report for WSSD prepared by IWA http://www.gdrc.org/uem/water/iwrm/1pager-01.html.
IWMI, 2006. Insights from the Comprehensive Assessment of Water Management in Agriculture. International Water Management Institute (IWMI), Colombo, Sri Lanka.
Jackson, R.B., E.G. Jobbagy, R. Avissar, S.B. Roy, D.J. Barrett, C.W. Cook, K.A. Farley, D.C. le Maitre, B.A. McCarl & B.C. Murray, 2005. Trading water for carbon with biological sequestration. Science 310: 1944–1947.
Jähnig, S.C., K. Brabec, A. Buffagni, S. Erba, A.W. Lorenz, T. Ofenböck, P. F. M. Verdonschot & D. Hering, 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47: 671-680.
Juszczak, R., A. Kędziora, J. Olejnik, 2007. Assessment of Water Retention Capacity of Small Ponds in Wyskoć Agricultural-Forest Catchment in Western Poland. Polish Journal of Environmental Studies 16: 685-695.
Kaplan, J.O., K.M. Krumhardt & N. Zimmermann, 2009. The prehistoric and preindustrial deforestation of Europe. Quaternary Science Reviews 28: 3016-3034.
Kelly, D.J., M.L. Bothwell & D.W. Schindler, 2003. Effects of solar ultraviolet radiation on stream benthic communities: An intersite comparison. Ecology 84:2724–2740.
Khan, G. D., M. Latif & S. Hassan, 2003. The role of controlled drainage under drought conditions in an irrigated area in NWFP, Pakistan. Irrigation and Drainage 52: 147-162.
Khan, V.M., R.M. Vilfand & P.O. Zavialov, 2004. Long-term variability of air temperature in the Aral Sea region. Journal of Marine Systems 47: 25–33.
Kilroy, G., C. Coxon, D. Daly, A. O'Connor, F. Dunne, P. Johnston, J., Ryan H. Moe & M. Craig, 2009. Monitoring the environmental supporting conditions

of groundwater dependent terrestrial ecosystems in Ireland. In: Quevauviller, P., A.-M. Fouillac, J. Grath & R. Ward (eds.), Groundwater Monitoring, Wiley, Chichester: 245 – 258.
Kotanen, J., P. Manninen, A. Petäjä-Ronkainen, A. Panula-Ontto-Suuronen, 2009. Yhteistyöllä parempaan vesienhoitoon. Vuoksen vesienhoitoalueen vesienhoitosuunnitelma vuoteen 2015.
Kuusemets, V., Ü. Mander, K. Lõhmus & M. Ivask, 2001. Nitrogen and phosphorus variation in shallow groundwater and assimilation in plants in complex riparian buffer zones. Water Science and Technology 44: 615–622.
Köhler, J., J. Hachoł & S. Hilt, 2010: Regulation of submersed macrophyte biomass in a temperate lowland river: Interactions between shading by bank vegetation, epiphyton and water turbidity. Aquatic Botany 92: 129–136.
Lal, R., 2008. Sequestration of atmospheric CO_2 in global carbon pools. Energy & Environmental Science 1: 86-100.
Lamm, F.R. & C.R. Camp, 2007. Subsurface drip irrigation. Developments in Agricultural Engineering 13: 473-551.
LeBlanc, A., 1999. Issues related to including forestry-based offsets in a GHG emissions trading system. Environmental Science & Policy 2: 199-206.
Lecerf, A. & J.S. Richardson, 2010. Litter decomposition can detect effects of high and moderate levels of forest disturbance on stream condition. Forest Ecology and Management 259: 2433–2443.
Lehner, B., Henrichs, T., Döll, P., Alcamo, J. (2001): EuroWasser – Model-based assessment of European water resources and hydrology in the face of global change. Kassel World Water Series 5, Center for Environmental Systems Research, University of Kassel.
Lenhart, T., N. Fohrer & H.-G. Frede, 2003. Effects of land use changes on the nutrient balance in mesoscale catchments. Physics and Chemistry of the Earth 28: 1301–1309.
Li, Z., F. Boyle & A. Reynolds, 2010. Rainwater harvesting and greywater treatment systems for domestic application in Ireland. Desalination 260: 1-8.
Li, X.-G., Y.-K. Li, F.-M. Li, Q. Ma, P.-L. Zhang & P. Yin, 2009. Changes in soil organic carbon, nutrients and aggregation after conversion of native desert soil into irrigated arable land. Soil & Tillage Research 104: 263–269.
LIFE, 2006. Living with water. 49 water projects in the Netherlands within the context of LIFE Environment (1992-2006).
Lin, C.Y., W.C. Chou & W.T. Lin, 2002. Modeling the width and placement of riparian vegetated buffer strips: a case study on the Chi-Jia-Wang stream. Taiwan Journal of Environmental Management 66:269–280.
Mack, S. K., J.W. Day, R. Lane, A. J. Englande, R. S. Reimers, J. Groby and G. C. Austin, 2010. Wetland Assimilation: A case study of climate change mitigation and adaptation in New Orleans. International Climate Change Adaptation Conference. Climate Adaptation Futures: Preparing for the unavoidable impacts of climate change, 29 June - 1 July 2010, Book of abstracts.
Makra, L, J. Mika & S. Horváth, 2005. 20th century variations of the soil moisture content in East-Hungary in connection with global warming Physics and Chemistry of the Earth, Parts A/B/C 30:181-186.

Mander, Ü., 2008. Riparian Zone Management and Restoration. Encyclopedia of Ecology, 2008, Pages 3044-3061.
Mander, Ü., Y. Hayakawa & V. Kuusemets, 2005. Purification processes, ecological functions, planning and design of reparian buffer zones in agricultural watersheds. Ecological Engineering 24: 421–432.
Mander,Ü. & K. Kimmel, 2008. Wetlands and riparian buffer zones in landscape functioning. In: Hong, S.-K., N. Nakagoshi, B.J. Fu & Y. Morimoto (eds.), Landscape ecological applications in man-influenced areas: Linking man and nature systems. Springer: 329-357.
Marchi, L., M. Borga, E. Preciso & E. Gaume, 2010. Characterisation of selected extreme flash floods in Europe and implications for flood risk management. Journal of Hydrology 394: 118-133.
Marttila, H. & B. Kløve, 2010. Managing runoff, water quality and erosion in peatland forestry by peak runoff control. Ecological Engineering 36: 900–911.
Marttila, H. & Kløve, B., 2009. Retention of sediment and nutrient loads with peak runoff control. Journal of Irrigation and Drainage Engineering 13: 210–216.
Maticic, B. 1999. The impact of agriculture on ground water quality in Slovenia: standards and strategy. Agricultural Water Management 40:235-247.
Maticic, B., 2004. Rain harvesting in karst regions in Slovenia. http://www.wg-crop.icidonline.org/116_late.pdf.
McAlister, T., Mitchell, G., Fletcher, T., Phillips, B., 2003. Modelling urban stormwater management systems. Chapter 13 in. In: Wong, T.H.F. (Ed.). Australian Runoff Quality, Draft. Engineers Australia, Sydney, Australia.
McClain, M.E., E.W. Boyer, C.L. Dent, S.E. Gergel, N.B. Grimm, P.M. Groffman, S.C. Hart, J.W. Harvey, C.A. Johnston, E. Emilio Mayorga, W.H. McDowell & G. Pinay, 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 2003, 6:301-12.
McCullough, D. A., J.M. Bartholow, H.I. Jager, R.L. Beschta, E.F. Cheslak, M.L. Deas, J.L. Ebersole, J.S. Foott, S.L. Johnson, K.R. Marine, M.G. Mesa, J.H. Petersen, Y. Souchon, K.F. Tiffan & W.A. Wurtsbaugh, 2009. Research in Thermal Biology: Burning Questions for Coldwater Stream Fishes. Reviews in Fisheries Science 17: 90–115
McGroddy, M.E., W.T. Baisden & L.O. Hedin, 2008. Stochiometry of hydrological C, N and P losses across climate and geology: an environmental matrix approach across New Zealand primary forests. Global Biogeochem Cycles 2008, 22: GB3005.
McKergow, L.A., D.M. Weaver, I.P. Prosser, R.B. Grayson & A.E.G. Reed, 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment. Western Australian Journal of Hydrology 270: 253–272.
Meals, D.W., 2001. Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. Water Science and Technology 43: 175–182.
Meleason, M.A. & J.M. Quinn, 2004. Influence of riparian buffer width on air temperature at Whangapoua Forest, Coromandel Peninsula, New Zealand. Forest Ecology and Management 191: 365–371.
Merrett, S. 2004. Integrated water resources management and the hydrosocial balance. Water international 29:148-157.

Merz, R. & G. Bloschl, 2009. A regional analysis of event runoff coefficients with respect to climate and catchment characteristics in Austria. Water Resources Research 45: W01405, 19 PP., doi: 10.1029/2008WR007163.
Montgomery, D., 2008. Dirt: The Erosion of Civilizations. University of California Press.
Montreuil, O. & P. Merot, 2006. Nitrogen removal in valley bottom wetlands: assessment in headwater catchments distributed throughout a large basin. Journal of Environmental Quality 35: 2113-2122.
Mooij, W.M., L.N. De Senerpont Domis, S. Hülsmann, 2008. The impact of climate warming on water temperature, timing of hatching and young-of-the-year growth of fish in shallow lakes in the Netherlands. Journal of Sea Research 60: 32-43.
Moore, R.D. & S.M. Wondzell, 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A Review. Journal of the American Water Resources Association 41:763-784.
Moore, R.D., D.L. Spittlehouse & A. Story, 2005. Riparian microclimate and stream temperature response to forest harvesting: a review. Journal of American Water Resources Association 41: 813–834.
Moran, M.S., T.R. Clarke, Y. Inoue & A. Vidal, 1994: Estimating crop water deficit using the relation between surface-air temperature and spectral vegetation index. Remote Sensing Environment 49: 246–263.
Moss, T. & J. Monstadt, 2008. Restoring Floodplains in Europe. Policy Contexts and Project Experiences, International Water Association, ISBN 1843390906.
Mulholland, P.J., A.M. Helton, G.C. Poole, R.O. Hall, S.K. Hamilton, B.J. Peterson, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, W.K. Dodds, S.E. Findlay, S.V. Gregory, N.B. Grimm, S.L. Johnson, W.H. McDowell, J.L. Meyer, H.M. Valett, J.R. Webster, C.P. Arango, J.J. Beaulieu, M.J. Bernot, A.J. Burgin, C.L. Crenshaw, L.T. Johnson, B.R. Niederlehner, J.M. O'Brien, J.D. Potter, R.W. Sheibley, D.J. Sobota et al., 2008. Stream denitrification across biomes and its response to anthropogenic nitrate loading. Nature 452: 202-205.
Nakamura, F. & H. Yamada, 2005. Effects of pasture development on the ecological functions of riparian forests in Hokkaido in northern Japan. Ecological Engineering 24: 539–550.
Neal, C., J. Hilton, A.J. Wade, M. Neal & H. Wickham, 2006. Chlorophyll-a in the rivers of eastern England. Science of Total Environment 365: 84-104.
Neuschulz, F. & J. Purps, 2003. Auenregeneration durch Deichrückverlegung – ein Naturschutzprojekt an der Elbe bei Lenzen mit Pilotfunktion für einen vorbeugenden Hochwasserschutz. Naturschutz und Landschaftspflege in Brandenburg 3/2003.
Niklaus, P.A. & C.Körner, 2004. Synthesis of a six-year study of calcareous grassland responses to in situ CO_2 enrichment. Ecological Monographs 74: 491-511.
Nishida, H., M. Konno, A. Ikeda & Y. Tokiwa, 2000. Microbial degradation of poly(p-dioxanone) I. Isolation of degrading microorganisms and microbial decomposition in pure culture. Polymer Degradation and Stability 68: 205-217.
Nixon, S. 2008. Summary of information received from Member States on best practices and approaches for a climate check of the first Programmes of

Measures. WRc Report to the Strategic Steering Group on Climate Change and Water. Version no. 2, 21 August 2008.
NL, 2009. 2009 - 2015 Nationaal Waterplan. 22 december 2009
NZWERF, 2004. On-Site Stormwater Management Manual. New Zealand. Water Environment Research Foundation, Wellington, New Zealand.
Nõges, P., R. Adrian, O. Anneville, L. Arvola, T. Blenckner, D. G. George, T. Jankowski, M. Järvinen, S. C. Maberly, J. Padisák, D. Straile, K. Teubner and G. Weyhenmeyer, 2010. The impact of variations in the climate on seasonal dynamics of phytoplankton. Ch. 14 in D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer, p. 253-274.
Nõges, T., P. Nõges & A.C. Cardoso. 2010. Climate change adaptation and mitigation strategies already in practice based on the 1st River Basin Management Plans of the EU Member States. JRC Scientific and Technical Reports. EUR 24663 EN. Luxembourg, Publications Office of the European Union, 127 pp. DOI 10.2788/83841
O'Connor, J.E. & J.E. Costa, 2004. Spatial distribution of the largest rainfall-runoff floods from basins between 2.6 and 26,000 km^2 in the United States and Puerto Rico. Water Resources Research 40. W01107.
O'Driscoll, M.A. & D.R. DeWalle, 2006. Stream–air temperature relations to classify stream–ground water interactions in a karst setting, central Pennsylvania, USA. Journal of Hydrology 329: 140–153.
Ogle, S.M., F.J. Breidt, M.D. Eve & K. Paustian, 2003. Uncertainty in estimating land use and management impacts on soil organic storage for US agricultural lands between 1982 and 1997. Global Change Biology 9: 1521–1542.
Olschewski, R., P.C. Benítez, G.H.J. de Koning & T. Schlichter, 2005. How attractive are forest carbon sinks? Economic insights into supply and demand of Certified Emission Reductions. Journal of Forest Economics 11: 77-94.
Osborne, L.L., P.B. Bayley & L.W. Higler (eds.), 1993. Lowland stream restoration: Theory and practice. – Freshwater Biology (special issue) 2: 187-342.
Ostle, N.J., P.E. Levy, C.D. Evans & P. Smith, 2009. UK land use and soil carbon sequestration. Land Use Policy 26, Supplement 1: S274-S283.
Oudin, L., V. Andreassian, J. Lerat & C. Michel, 2008. Has land cover a significant impact on mean annual streamflow? An international assessment using 1508 catchments. Journal of Hydrology 357: 303-316.
Parish F, Sirin A, Charman D, Joosten H, Minaeva T, Silvius M (eds) (2008) Assessment on peatlands, biodiversity and climate change. Global Environment Centre, Kuala Lumpur and Wetlands International Wageningen, 179 p.
Parkes, M.W., K.E. Morrison, M.J. Bunch, L.K. Hallström, R.C. Neudoerffer, H.D. Venema & D.Waltner-Toews, 2010. Towards integrated governance for water, health and social–ecological systems: The watershed governance prism. Global Environmental Change 20: 693-704.
Perkins, D.W. & M.L. Hunter Jr., 2006. Use of amphibians to define riparian zones of headwater streams. Canadian Journal of Forest Research 36: 2124–2130.

Peterjohn, W.T., Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 1984, 65:1466-75.
Peterson, S.A. 1981. Sediment removal as a Lake Restoration technique. Corvallis Env. Res. Lab., USEPA. EPA-600/3-81-013.56 p.
Pickering, K.T. & L.A. Owen 1997. An introduction to global environmental issues. Routledge.
Pilliod, D.S., R.B. Bury, E.J. Hyde, C.A. Pearl & P.S. Corn, 2003. Fire and amphibians in North America. Forest Ecology and Management 178: 163–181.
Pinay, G. and D.M. Hannah, 2009. Evaluation of global change impacts on diffuse pollution. F1000 Biology Reports 82, (doi:10.3410/B1-82).
Pivot, J.-M., E. Josien & P. Martin, 2002. Farms adaptation to changes in flood risk: a management approach. Journal of Hydrology 267: 12–25.
Poole, G.C. & C.H. Berman, 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27: 787–802.
Poulard, C., M. Lafont, A. Lenar-Matyas & M. Łapuszek, 2008. Flood mitigation designs with respect to river ecosystem functions—A problem oriented conceptual approach. Ecological Engineering 36: 69-77.
Powers, J.S. & W.H. Schlesinger, 2002. Relationships among soil carbon distributions and biophysical factors at nested spatial scales in rain forests of northeastern Costa Rica. Geoderma 109: 165-190.
Powlson, D.S., D.G. Christian, P. Falloon & P. Smith, 2002. Biofuel crops: their potential contribution to decreased fossil carbon emissions and additional environmental benefits. Aspects of Applied Biology 65:289–94.
Prabhakar, S.V.R.K. & M. Elder, 2009. Biofuels and resource use efficiency in developing Asia: Back to basics. Applied Energy 86, Supplement 1: S30-S36.
Prudhomme, C. & H. Davies, 2009. Assessing uncertainties in climate change impact analyses on the river flow regimes in the UK. Part 1: baseline climate. Climate Change 93:177-95.
Puustinen, M., S. Tattari, J. Koskiaho & J. Linjama, 2007. Influence of seasonal and annual hydrological variations on erosion and phosphorus transport from arable areas in Finland. Soil & Tillage Research 93: 44–55.
Querner, E. P & H. A. J. van Lanen, 2001. Impact assessment of drought mitigation measures in two adjacent Dutch basins using simulation modeling. Journal of Hydrology 252: 51-64.
Quinn, J.M. & A.E. Wright-Stow, 2008. Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest Ecology and Management 256: 2101-2109.
Quinn, J.M., A.B. Cooper, M.J. Stroud & G.P. Burrell, 1997. Shade effects on stream periphyton and invertebrates: an experiment in streamside channels. New Zealand Journal of Marine and Freshwater Research 31: 129–146.
Rashin, E.B., C.J. Clishe, A.T. Loch & J.M Bell, 2006. Effectiveness of timber harvest practices for controlling sediment related water quality impacts. Journal of American Water Resources Association 42: 1307–1327.

Rhoades, R.E., 1998. Participatory watershed research and management: where the shadow falls. Gatekeeper Series, 81, Landon: International Institute for Environment and Development.
Richardson, D.M., P.M. Holmes, K.J. Esler, S.M. Galatowitsch, J.C. Stromberg, S.P. Kirkman, P. Pyšek & R.J. Hobbs, 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Diversity and Distributions 13: 126-139.
Ripl, W. & C. Hildmann, 2000. Dissolved load transported by rivers as an indicator of landscape sustainability. Ecological Engineering 14: 373–387.
Roehm, C.L., 2005. Respiration in wetland ecosystems. In: del Giorgio, P.A., Williams, P.J., le B., Eds. Respiration in Aquatic systems. Oxford: Oxford University Press. pp 83–102.
Rosenzweig, C. & F.N. Tubiello, 2007. Adaptation and mitigation strategies in agriculture: an analysis of potential synergies. Mitigation and Adaptation Strategies of Global Change 12: 855–873.
Rossi, L. & R.E. Hari, 2007. Screening procedure to assess the impact of urban stormwater temperature to populations of brown trout in receiving water. Integrated Environmental Assessment and Management 3: 383–392.
Roulet, N.T., 2000. Peatlands, carbon storage, greenhouse gases, and the Kyoto protocol: prospects and significance for Canada. Wetlands 20: 605–615.
Räisanen, J., U. Hansson, A. Ullerstig, R. Döscher, L. P. Graham, C. Jones, H. E. M. Meier, P. Samuelsson & U. Willén. 2004. European climate in the late twenty-first century: regional simulations with two driving global models and two forcing scenarios. Climate Dynamics 22:13–31.
Rykken, J.J., A.R. Moldenke & D.H. Olson, 2007. Headwater Riparian Forest-Floor Invertebrate Communities Associated with Alternative Forest Management Practices. Ecological Applications 17: 1168-1183.
Saarinen, T., K.-M. Vuori, E. Alasaarela & B. Kløve, 2010. Long-term trends and variation of acidity, CODMn and colour in coastal rivers of Western Finland in relation to climate and hydrology. Science of The Total Environment 408: 5019-5027.
Sabater, S., A. Butturini, J.C.Clement, T.P. Burt, D. Dowrick, M. Hefting, V. Maître, G. Pinay, C. Postolache, M. Rzepecki, F. Sabater, 2003. Nitrogen removal by riparian buffers under various N loads along a European climatic gradient: patterns and factors of variation. Ecosystems 2003, 6:20-30.
Salvetti, R., A. Azzellino & R. Vismara, 2006. Diffuse source apportionment of the Po river eutrophying load to the Adriatic sea: Assessment of Lombardy contribution to Po river nutrient load apportionment by means of an integrated modelling approach. Chemosphere 65: 2168-2177.
Sanchez, C.A., D. Zerihun & K.L. Farrell-Poe, 2009. Management guidelines for efficient irrigation of vegetables using closed-end level furrows. Agricultural Water Management 96: 43-52.
Sarkar, S.K., M. Saha, H. Takada, A. Bhattacharya, P. Mishra & B. Bhattacharya, 2007. Water quality management in the lower stretch of the river Ganges, east coast of India: an approach through environmental education. Journal of Cleaner Production 15: 1559-1567.
Sarr, D.A., 2002. Riparian livestock exclosure research in the western United States: A critique and some recommendations. Environmental Management 30: 516-526.

Schilling, K.E., K.-S. Chan, H. Liu, Y.-K. Zhang, 2010. Quantifying the effect of land use land cover change on increasing discharge in the Upper Mississippi River. Journal of Hydrology 387: 343–345.
Schipper, L.A., W.T. Baisden, R.L. Parfitt, C. Ross, J.J. Claydon & A. Greg, 2007. Large losses of soil C and N from soil profiles under pasture in New Zealand during the past 20 years. Global Change Biology 13: 1138-1144.
Schoorl, J.M. & A. Veldkamp, 2001. Linking land use and landscape process modelling: a case study for the Álora region (south Spain). Agriculture, Ecosystems and Environment 85: 281–292.
Schueler, T. R. 2000. The Importance of Imperviousness. The Practice of Watershed Protection. Center for Watershed Protection. Ellicott City, MD.
SEPA, 2007. Significant water management issues in the Scotland river basin district. Scottish Executive Central Research Unit (2002) Climate Change: Flood Occurrence Review.
SEPA, 2009. The river basin management plan for the Scotland river basin district 2009–2015. Chapter 3: Achieving our environmental objectives.
Sheehy Skeffington, M., J. Moran, A. O'Connor, E. Regan, C.E. Coxon, N.E. Scott & M. Gormally, 2006. Turloughs – Ireland's unique wetland habitat. Biological Conservation 133: 265-290.
SIC, 2009. Scientific-Information Center of Interstate Commission for Water Coordination (SIC ICWC), Press release No. 1 (159) April 2009: http://sic.icwc-aral.uz/releases/eng/159.htm.
Sivapalan, M., G. Blöschl, R. Merz & D.Gutknecht, 2005. Linking flood frequency to long-term water balance: Incorporating effects of seasonality. Water Resources Research 41. W06012.
Skaggs, R. W., Breve, M. A. & Gilliam, J. W. 1994. Hydrologic and water quality impacts of agricultural drainage. Critical Reviews in Environmental Science and Technology 24: 1-32.
Sleator, R.D., 2010. The human superorganism – Of microbes and men. Medical Hypotheses 74: 214-215.
Smakhtin, V.U., 2001. Low flow hydrology: a review. Journal of Hydrology 240: 147–186.
Smith, S.V., W.H. Renwick, J.D. Bartley & R.W. Buddemeier, 2002. Distribution and significance of small, artificial water bodies across the United States landscape. Science of Total Environment 299: 21– 36.
Somper, C. 2005. The Parrett Catchment Project. South West Sustainable Land Use Initiative. Forum for the Future, 9
Sophocleous, M., 2002. Interactions between groundwater and surface water: the state of science. Hydrogeology Journal 10: 52–67.
Spatial Planning Key Decision 'Room for the River', 2006. Investing in the safety and vitality of the Dutch river basin region. Ministry of Transport, Public Works and Water Management.
Stephenson, N. L. 1998. Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of biogeography 25: 855-870.
Sternberg, R., 2008.Hydropower: Dimensions of social and environmental coexistence. Renewable and Sustainable Energy Reviews 12:1588-1621.
Stomberg, J., R. Tiller & B. Richter, 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: The San Pedro, Arizona. Ecological Applications 6: 113-131.

Sweeney, B.W., T.L. Bott, J.K. Jackson. J.D. Newbold, L.J. Standley, W.C. Hession & R.J. Horwitz, 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. PNAS 101:14132–14137.
Zammit, C., M. Sivapalan, P. Kelsey, N.R. Viney, 2005. Modelling the effects of land-use modifications to control nutrient loads from an agricultural catchment in Western Australia. Ecological Modelling 187: 60–70.
Zencich, S.J., R.H. Froend, J.T. Turner & V. Gailitis, 2002: Influence of groundwater depth on the seasonal sources of water accessed by Banksia tree species on a shallow, sandy coastal aquifer. Oecologia 131: 8–19.
Zhang, M.-K., L.-P. Wang & Z.-L. He, 2007. Spatial and temporal variation of nitrogen exported by runoff from sandy agricultural soils. Journal of Environmental Sciences 19: 1086–1092.
Zoppou, C., 2001. Review of urban storm water models. Environmental Modelling and Software 16, 195e231.
TEEB, 2009. The Economics of Ecosystems and Biodiversity for National and International Policymakers, website accessed December 2010. (http://www.teebweb.org/ForPolicymakers/tabid/1019/language/en-US/Default.aspx).
Tian, Y., Y. He & L. Guo, 2009. Soil water carrying capacity of vegetation in the northeast of Ulan Buh Desert, China. Frontiers of Forestry in China 4: 309–316.
Todd, A.S., M.A. Coleman, A.M. Konowal, M.K. May, S. Johnson, N.K.M. Vieira & J.F. Saunders, 2008. Development of New Water Temperature Criteria to Protect Colorado's Fisheries. Fisheries 33: 433-443.
Tomlinson, M. & A. Boulton, 2008. Subsurface Groundwater Dependent Ecosystems: a review of their biodiversity, ecological processes and ecosystem services. Waterlines Occasional Paper No 8.
Trabucco, A., R.J. Zomer, D.A. Bossio, O. van Straaten & L.V. Verchot, 2008. Climate change mitigation through afforestation/reforestation: A global analysis of hydrologic impacts with four case studies. Agriculture, Ecosystems and Environment 126: 81–97.
Trowbridge, W.B., 2007. The role of stochasticity and priority effects in floodplain restoration. Ecological Applications 17: 1312-1324.
Trumper, K., M. Bertzky, B. Dickson, G. van der Heijden, M. Jenkins & P. Manning, P., 2009, The Natural Fix? The role of ecosystems in climate mitigation — A UNEP rapid response assessment, United Nations Environment Programme, UNEP-WCMC, Cambridge, United Kingdom
Tunowski, J. 2009. Changes in zooplankton abundance and community structure in the cooling channel system of the Konin and Pątnów power plants. Archives of Polish Fisheries 17: 279-289.
Turbé, A., A. De Toni, B. Benito, P. Lavelle, N. Ruiz, W.H. Van der Putten, E. Labouze & S. Mudgal, 2010. Soil biodiversity: functions, threats and tools for policymakers, Bio-Intelligence Service, IRD, and NIOO, Report for European Commission, DG Environment. (http://ec.europa.eu/environment/soil/pdf/biodiversity_report.pdf).
Turnock, D., 2002. Ecoregion-based conservation in the Carpathians and the land-use implications. Land Use Policy 19: 47-63.
Twery, M.J. & J.W. Hornbeck, 2001. Incorporating water goals into forest management decisions at a local level. Forest Ecology and Management 143: 87-93.

Urbanová, Z., T. Picek, J. Bárta, 2010. Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. Ecological Engineering (in press) doi:10.1016/j.ecoleng.2010.07.012.
US EPA 2001. Functions and values of wetlands. Factsheet EPA 843-F-01-002c.
van den Brink, H.W., G.P. Können, J.D. Opsteegh, G.J. van Oldenborgh & G. Burgers, 2005. Estimating return periods of extreme events from ECMWF seasonal forecast ensembles. International Journal of Climatology 25: 1345–1354.
van Dijk, A.I.J.M., P.B. Hairsinea, J.P. Arancibia and T.I. Dowling, 2007. Reforestation, water availability and stream salinity: A multi-scale analysis in the Murray-Darling Basin, Australia. Forest Ecology and Management 251: 94-109.
Vandewalle, M., M.T., Sykes, P.A., Harrison, G.W., Luck, , P., Berry R.Bugter, , T.P., Dawson, Feld, C.K., R.Harrington, J.R.,, Haslett, D Hering,., K.B., Jones, R.Jongman, , S Lavorel,., P., Martins da Silva, M.Moora, , J.Paterson, , M.D.A., Rounsevell, , L., Sandin J., Settele, J.P., Sousa, and M. Zobel, 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), website accessed December 2010. (http://www.rubicode.net/rubicode/RUBICODE_Review_on_Ecosystem_Services.pdf).
Wang, F. C. & A. R. Overman, 1981. Impacts of surface drainage on ground water hydraulics. Journal of the American Water Resources Association 17: 971-977.
Wang, G., H. Hu & T. Li, 2009. The influence of freeze–thaw cycles of active soil layer on surface runoff in a permafrost watershed. Journal of Hydrology 375: 438–449.
Wang, Q & H. Takashi, 1999. A Land Surface Water Deficit Model for an Arid and Semiarid Region: Impact of Desertification on the Water Deficit Status in the Loess Plateau, China. Journal of Climate 12: 244-257.
Wantzen, K.M. & W.J. Junk, 2008. Riparian Wetlands. Encyclopedia of Ecology, Elsevier: 3035-3044.
Veen, P., R. Jefferson, J. de Smidt & J. van der Straaten, (eds), 2009. Grasslands in Europe of high nature value, KNNV Publishing.
Wesström, I., I. Messing, H. Linnér & J. Lindström, 2001. Controlled drainage — effects on drain outflow and water quality. Agricultural Water Management 47: 85-100.
WFD, 2000. Directive 2000/60/EC of the European Parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. Official Journal of the European Communities L 327: 1–72.
Wick, L., G. Lemcke & M. Sturm, 2003. Evidence of Lateglacial and Holocene climatic change and human impact in eastern Anatolia: high-resolution pollen, charcoal, isotopic and geochemical records from the laminated sediments of Lake Van, Turkey. The Holocene 13: 665–675.
Wilby, R.L., H. Orr, G. Watts, R.W. Battarbee, P.M. Berry, R. Chadd, S.J. Dugdale, M.J. Dunbar, J.A. Elliott, C. Extence, D.M. Hannah, N. Holmes,

A.C. Johnson, B. Knights, N.J. Milner, S.J. Ormerod, D. Solomon, R. Timlett, P.J. Whitehead & P.J. Wood, 2010. Evidence needed to manage freshwater ecosystems in a changing climate: Turning adaptation principles into practice. Science of the Total Environment 408: 4150–4164.

Wilk, J. and Wittgren, H.B. (eds). 2009. Adapting Water Management to Climate Change. Swedish Water House Policy Brief Nr. 7. SIWI.

Wilkerson, E., J.M. Hagan, D. Siegel & A.A. Whitman, 2006. The Effectiveness of Different Buffer Widths for Protecting Headwater Stream Temperature in Maine. Forest Science 52: 221-231.

Winpenny, J. 1994. Managing water as an economic resource. London: Routledge for the Overseas Development Institute. World Bank. 1991.

Woodward, K.B., C.S. Fellows, C.L. Conway & H.M. Hunter, 2009. Nitrate removal, denitrification and nitrous oxide production in the riparian zone of an ephemeral stream. Soil Biology and Biochemistry 41: 671-680.

WSDE, 2005. Stormwater Management Manual for Western Washington. Washington State Department of Ecology. http://www.ecy.wa.gov/pubs/0510029.pdf

European Commission

EUR 24682 EU – Joint Research Centre – Institute for Environment and Sustainability
Title: **Review of published climate change adaptation and mitigation measures related with water**

Author(s): **Tiina Nõges, Peeter Nõges, Ana Cristina Cardoso**

Luxembourg: Publications Office of the European Union
2010 – 127 pp. – 21 x 29.7 cm
EUR – Scientific and Technical Research series – ISSN 1018-5593
ISBN 978-92-79-18984-5
doi:10.2788/18203

Abstract
Stronger manifestation of climate change impact on global water cycle, water resources, and aquatic ecosystems has given a strong impetus to the development of adaptation measures in water management. The present report gives an insight to potential and planned water related measures tackling climate change causes and consequences, which have been included in the Member States River Basin Management Plans, published in various reports and scientific literature mostly within the last decade. The database of about 450 measures analysed in this report and given in a separate Annex as an Excel spreadsheet, constitutes the most important part of this deliverable. In the context of this report, measures are defined as practical steps or actions taken to (i) reduce the sources or enhance the sinks of greenhouse gases, (ii) to decrease the vulnerability of water resources and aquatic ecosystems to climate change, or (iii) enhance the knowledge base on climate-water relationships and increase the societal capacity to take right decisions on this matter. By strategic approach, the measures belong either to planned adaptation, which specifically focuses on climate change and variability, and autonomous adaptation, which goals are not specifically climate related, but have an added value in improving resilience to climate change. Separate chapters are dedicated to each of the five specific adaptation strategies addressed in the REFRESH Project. The present report is of relevance to the 7th EU Framework Programme, Theme 6 (Environment including Climate Change) project REFRESH (Adaptive strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems, Contract No.: 244121), to JRC Thematic Area 3 (Sustainable management of natural resources) foci on CC, to the European Clearing House mechanism on CC, and to the EC Blueprint on Water.

How to obtain EU publications

Our priced publications are available from EU Bookshop (http://bookshop.europa.eu), where you can place an order with the sales agent of your choice.

The Publications Office has a worldwide network of sales agents. You can obtain their contact details by sending a fax to (352) 29 29-42758.

The mission of the JRC is to provide customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies. As a service of the European Commission, the JRC functions as a reference centre of science and technology for the Union. Close to the policy-making process, it serves the common interest of the Member States, while being independent of special interests, whether private or national.

LB-NA-24682-EN-C

EUROPEAN COMMISSION

Publications Office

ISBN 978-92-79-18984-5

9 789279 189845