The CLIMSAVE Project

Climate Change Integrated Assessment Methodology for Cross-Sectoral Adaptation and Vulnerability in Europe

Assessing Cross-Sectoral Adaptation and Mitigation Measures

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1. Introduction to adaptation and mitigation

The inter-relationship between adaptation and mitigation is an issue that has received increasing attention, both from a research and policy perspective due to its importance for decision-making and policy formulation. In the past, there was a dichotomy between mitigation and adaptation; they were thought of as separate issues, and hence interactions between the two were largely ignored (Biesbroek *et al.*, 2009). However, the two are inherently linked, for example, a high level of mitigation would require less adaptation and conversely if we adapt sufficiently, there is a possible reduced need for mitigation (Wilbanks *et al.*, 2007; Biesbreok *et al.*, 2009; van Vuuren *et al.*, 2011).

The IPCC included a new chapter on this subject in its Fourth Assessment Report and commented that there was a small but growing literature on this matter (Klein *et al.*, 2007). The chapter examined four types of relationship: (i) adaptation actions that have consequences for mitigation; (ii) mitigation actions that have consequences for adaptation; (iii) decisions that include trade-offs or synergies between adaptation and mitigation; and (iv) processes which have consequences for both adaptation and mitigation. The significance of adaptation and mitigation measures in different sectors for biodiversity was discussed by the Convention on Biological Diversity in their review of the interactions between biodiversity and adaptation and mitigation (Secretariat of the Convention on Biological Diversity, 2009). The inter-relationship also was acknowledged in the EU White Paper on Adaptation to Climate Change (COM (2009), 147 final) that stated the need to exploit synergies between mitigation and adaptation efforts. Since then, many studies have examined the interrelationship between adaptation and mitigation, and further highlighted the need for such integration in climate policy decisions (e.g. Wilbanks & Sathaye, 2007; Grayling, 2009; Pizarro, 2009; Fankhauser & Burton, 2011).

The interrelationship between adaptation and mitigation is however complex, with differences for planning in terms of the spatial, temporal, and administrative scales (see Beisbroek *et al.*, 2009 for discussion). Also there was a tendency for adaptation and mitigation to concern contrasting sectors, such that adaptation focused on sectors vulnerable to climatic change whereas mitigation mostly was undertaken by the energy, transport and industry sectors (Huq & Grubb, 2007). Increasingly, however, both in practice and policy there is recognition that there is a need for them to be addressed by all sectors. The interrelationships need to be well understood to maximise potential synergies, avoid conflicts, and carefully consider trade-offs (Tol, 2005; Harmin & Gurran, 2009; Smith & Oleson, 2010; VijayaVenkataRaman *et al.*, 2012). This can only be achieved by examining the issue using a holistic approach (Walsh *et al.*, 2010; Harry & Morad, 2013), and in terms of urban environments, research shows that spatial planning and integrated city models can be used to provide a framework to examine both adaptation and mitigation (Biesbroek *et al.*, 2009; Viguié & Hallegatte, 2012). Further research to improve understanding of the links between measures to reduce the risk of climate change would greatly improve policy, as win-win solutions are much more efficient than those with adverse affects (Laukkonen *et al.*, 2009; Walsh *et al.*, 2010; Smith, 2012; Viguié & Hallegatte, 2012). The importance of creating combined frameworks to assess climate change strategies is therefore essential (van Vuuren et al., 2011; Viguié & Hallegatte, 2012); there being no place for adaptation and mitigation dichotomy in future climate policy (Bosello *et al.*, 2013).

This review examines the adaptation and mitigation measures in each sector, as means to identifying those that might be relevant to modelling and understanding potential adaptation responses for the CLIMSAVE Integrated Assessment Platform (IAP), before considering their cross-sectoral interactions. It then explores the synergies and conflicts that may occur between adaptation and mitigation measures, as well as possible trade-offs, before considering the spatial and temporal scale of their implementation and the role of different levels of governance and other environmental and socio-economic impacts.

2. Methodology

The aim of the systematic search was to identify 25 relevant papers for each of the adaptation and mitigation options. The systematic search approach consisted of three main of stages: (i) generation of keywords; (ii) systematic search; and (iii) data extraction.

2.1 Generation of keywords

The MACIS report (Berry *et al.*, 2008a), which examined adaptation and mitigation responses to climate change for different sectors, including the six CLIMSAVE sectors of agriculture, forestry, biodiversity, the built environment, rivers, and coasts provided the starting point for this process. It enabled the identification of key responses and words associated with them for input into the systematic search. For agriculture, the English literature was searched, with a focus on identifying papers relating to Europe, while our Chinese partner searched the Chinese literature, the aim being to provide a comparison of adaptation and mitigation responses in the two areas.

It is important to recognise that much of the literature concerning climate change adopts different definitions for mitigation and adaptation, sometimes using them interchangeably. For CLIMSAVE, as in the MACIS report, the term mitigation included any actions seeking a net reduction of greenhouse gas (GHG) emissions, but also concerned the protection and promotion of carbon sinks, through land use and habitat management. Adaptation was defined as an action which avoids the unwanted impacts of climate change, and can also be a means of maintaining or restoring ecosystem resilience to single or multiple stresses (Convention on Biological Diversity, 2005). The MACIS Report contains a number of key adaptation and mitigation terms for each sector and these were extracted into a table, as shown below (Table 1), with additional terms being added where necessary. Alternative spellings, for example, dike and dyke, salt marsh and saltmarsh, were included in order to maximise the search success. The subject column of the table relates to what will be impacted by climate change. This can be, for example, an environment or a group of people, and was used to refine the search results when the adaptation term produced a large number of hits.

Table 1: Search keywords for the coastal sector.

Subject	Adaptation intervention	Mitigation intervention
Salt marsh	Dikes Dyke	Carbon storage
Estuaries	Beach nourishment	Wetland creation
Coastal wetlands	Embankment	Carbon sequestration
Coastal grazing marshland	Managed retreat	Carbon capture and storage
Intertidal wetlands	Managed realignment	

2.2 Systematic search

This was conducted using the online search database SciVerse Scopus, available at www.scopus.com. The search was conducted using the options "Article Title, Abstract, Keywords", all dates and all Subject Areas. There were two main stages to this part of the process, the first concerned adaptation (Figure 1) and the second, mitigation (Figure 2). The first step in the systematic search approach was to enter each adaptation intervention separately into the search database e.g. *"managed retreat"*. Some terms were quite specific, producing only a small number of hits, e.g. *"de-embankment"* (9), while other terms resulted in thousands of hits e.g. *"coastal engineering"* (9,409). If the adaptation term alone produced <100 hits, it was used as a standalone search term. If the adaptation term produced a large number of hits (≥100), then the search was refined by combining this adaptation term with each relevant subject term e.g. *dikes AND "salt marsh"; dikes AND estuaries*. Where there are two words in the search term e.g. "carbon storage", they were put in quotation marks in order to constrain the number of hits.

It is important to note that not all combinations of adaptation and subject terms were deemed relevant. Beach nourishment, for example, is unlikely to have any effect on either estuaries or coastal grazing marshland, and therefore it would be inappropriate to search for this combination of terms.

Articles of high relevance were those that contained a case-study example of the adaptation intervention, quantitative results, details on synergies, antagonisms, and trade-offs associated with the intervention. If the search produced <25 relevant hits, intelligent search approaches of snowballing and reverse snowballing were used. The former is where the reference lists of relevant articles are searched for secondary references which may be relevant and the latter uses citations of relevant articles as a means of searching for new articles.

As far as possible, articles relevant to Europe were used, but sometimes, where there was good evidence of the effects of adaptation and mitigation actions from other parts of the world then these were included.

The second step in the systematic search approach was to enter each mitigation term into the search database e.g. *"carbon storage"* and to repeat the adaptation methodology.

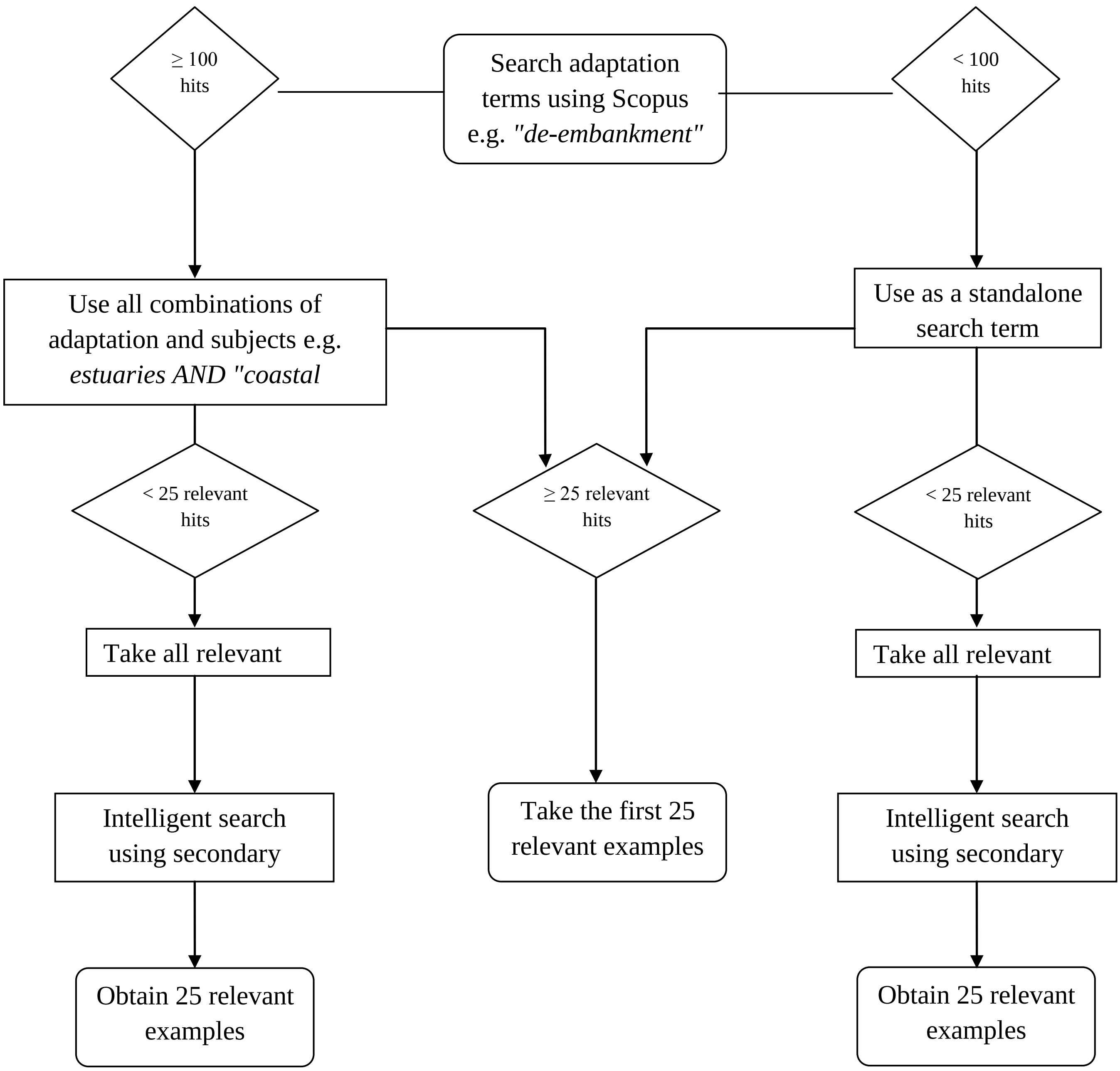

Figure 1: Search Process for adaptation.

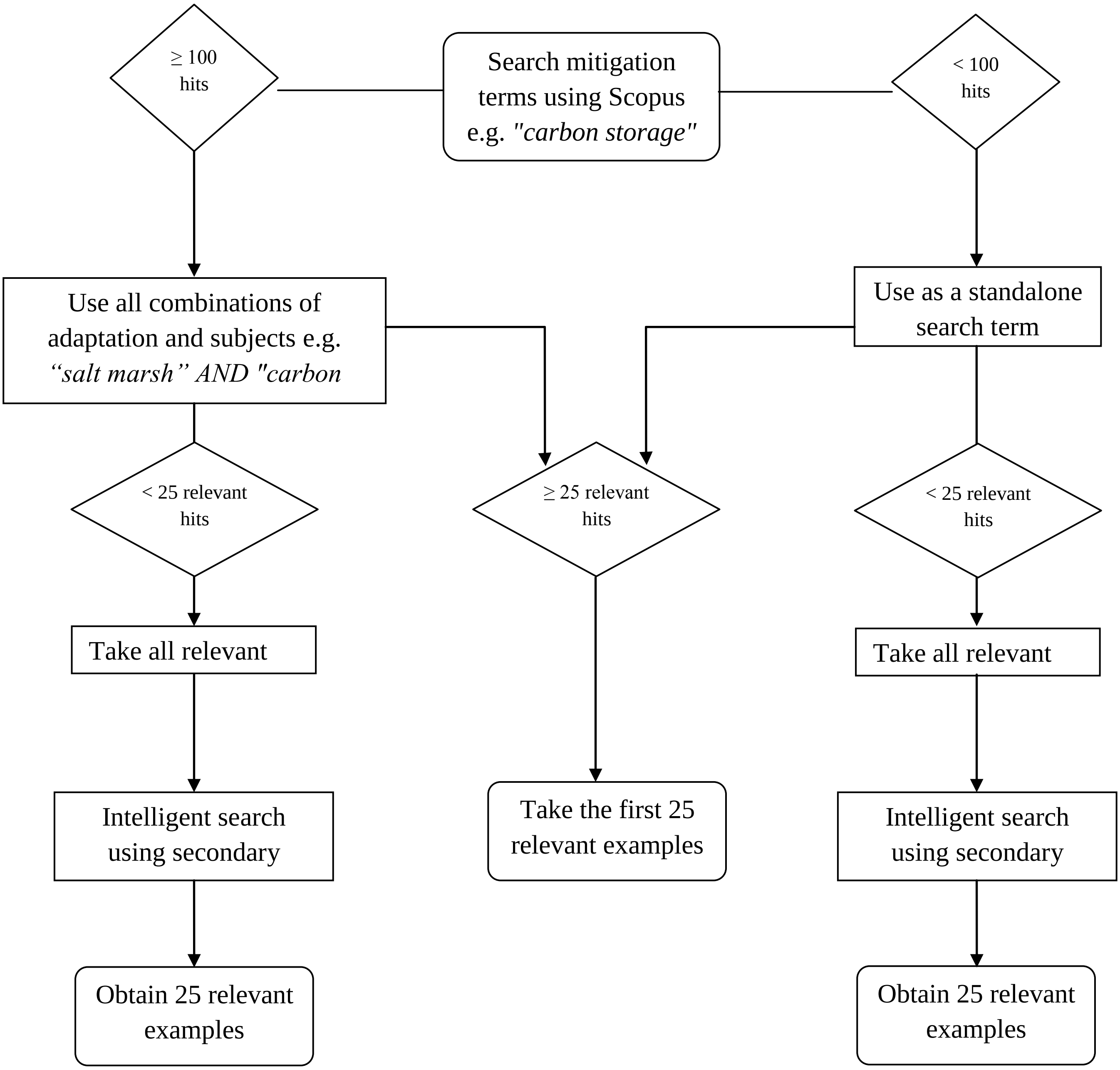

Figure 2: Search process for mitigation.

2.3 Data extraction

The data from relevant articles were extracted into a table which included columns relating to the scale of the project, governance, the actors involved, and any synergies and trade-offs. This method of data extraction helped highlight cross-sectoral linkages within and between mitigation and adaptation, as well as any synergistic or antagonistic interactions.

3. Adaptation options by sector

3.1 Agriculture

Agricultural adaptation to, and for, climate change includes changes of crop species or variety to cope with changing conditions, the wider use of technologies to harvest water, water management (e.g. to prevent water logging), alterations to timing of crop activities, diversification of income, improvement and effectiveness of integrated farm or crop management (e.g. pest control) and the use of seasonal forecasting to reduce production risk. Farmers may also adapt autonomously to changing conditions (Easterling *et al.*, 2007).

Adaptation is an important response for agriculture to address the potential impacts of climate change, but it also has a large contribution to make to mitigation. There are a range of possible short tactical and longer-term strategic adaptation options, which may be inter-linked, and it is important to ensure the former do not conflict with the latter (e.g. Howden *et al.*, 2007). This section focuses on adaptation and mitigation actions in Europe and China of particular relevance to the agricultural model in CLIMSAVE and which interacts with other sectors. These two regions will be discussed separately.

Based on the literature review, the main agricultural adaptation options in Europe were related to changing the timing of crop operations, using different cultivars and water management. The other adaptation options, such as minimum tillage, shade trees and drainage measures, produced very few hits and even using snowballing it was difficult to find additional papers and so attention was focused on the options for which there was good evidence.

3.1.1 Agriculture in Europe

<u>Spring and winter cropping</u>

Little research was found on switching from spring to winter cropping or vice versa as a means of adapting to unfavourable climatic conditions, but more research had been undertaken on adapting to the constraints on growth of spring and winter crops through changes in sowing dates or use of cultivars with different growing seasons (see below for a greater discussion on breeding). These can be effective, low-cost adaptation options to take advantage of changes in growing season or to avoid crop exposure to adverse climate (Wolfe *et al.*, 2008). An early study on optimising land use in Central Europe showed that the area of winter wheat, maize and vegetables could increase, while that of spring wheat, barley and potato could decrease (Parry *et al.*, 1988). More recently, Peltonen-Sainio *et al.* (2009) have suggested that in Finland climate change could result in autumn crops, such as winter wheat and rye being able to be grown in all arable areas, while triticale which currently only has limited overwintering success could become a major crop. They suggested that winter cereals with adequate overwintering capacity could replace spring ones as they have a greater ability to avoid summer drought and have future higher yield potential, such that by the 2050s winter wheat between 60 ^{o}N and 63^{o}N could yield 8.5 to 9.2 t ha^{-1} compared with 5.9 to 6.7 t ha^{-1} for spring wheat. New crops or new winter cultivars in Finland could include barley, oat, turnip and oilseed rape. A study of three crop rotations in Denmark, however, showed that including more spring cereals and catch crops in the rotations helped to offset the effects of climate change (Olesen *et al.*, 2004). Easterling *et al.* (2001) examined how the resolution of climate scenarios for the Great Plains, US, affected climate impacts on yields and adaptation strategies. They found that the different resolutions resulted in varying adaptation options, especially for maize and soybeans, which are more responsive than wheat to seasonal variations in precipitation. While Southworth *et al.* (2002) found that in the same region switching from maize (a C4 crop) to wheat (a C3 crop) could lead to increased growth and tolerance of high temperatures, as the latter could take more advantage of the higher CO_2 levels.

The adaptations undertaken are likely to be dependent on the projected climate change and cultivars available. Modelling for central Europe suggests that, while the frequency of suitable spring sowing days may increase, there could be greater variability in conditions which limit sowing, leading to higher inter-annual variability in crop yields (Trnka *et al.*, 2010). Earlier sowing, therefore, may not be possible in wet late winters/early springs and the recurrence of such weather might lead to winter crops being preferred, which are also better able to withstand spring drought stress events (Trnka *et al.*, 2010).

<u>*Spring crops*</u>

In Europe, for spring crops planting earlier with long season cultivars to take advantage of the extended growing season is an important short-term adaptation, which should increase yields providing there is adequate water available and the risk of heat stress is low, otherwise planting earlier with a short-season cultivar is the best response (Tubiello *et al.*, 2000; Olesen and Bindi, 2002; Adams *et al.*, 2003). Winter cereals need to have reached a specific growth stage before the onset of winter to ensure winter survival, and they are often sown when temperatures approach the time when vernalization is most effective. This may mean later sowings in northern Europe under climatic warming (Harrison *et al.*, 2000; Olesen *et al.*, 2000).

The need for adaptation and the forms of adaptation are also crop dependent (Tubiello *et al.*, 2002; Olesen 2004). For example, irrigated spring wheat on the Great Plains, US showed increases in yields under all scenarios, so that adaptation was not necessary, while for maize early planting could offset projected yield decreases and for potatoes it did little to counter the negative temperature effects (Tubiello *et al.*, 2002). Sowing trials in the north east of Western Australia of wheat cultivars with different developmental pattern and maturity dates showed that sowing between mid-May and early June produced the highest yields, but if sowing early, medium-long season cultivars generally had the better yields, while yields were best with short-season cultivars if late sowing (Kerr *et al.*, 1992).

For spring crops, earlier sowing dates often bring benefits in terms of increased yields, as they could allow crop growth during a period when adequate water is available (Alexandrov *et al.*, 2002). Moriondo *et al.* (2010) found that earlier sowing times could lead on average to a 5% decrease in drought stress (8%, 9% and 3% for sunflower, soybean and spring wheat respectively) and 8%, 9% and 3% for heat stress for sunflower, soybean and spring wheat respectively. A delay in sowing time led to the opposite results, although durum wheat seemed to be little affected by sowing date changes. Research into safflower sowing dates in Lebanon also found that earlier spring sowing had several benefits, including on yields (Yau, 2007), while Cuculeanu *et al.* (1999) when modelling maize in Romania found that later sowing dates could lead to an increase in yields.

Earlier sowing dates, combined with long season cultivars of spring crops, will increase the growing season, increasing yields if adequate water is available and the risk of heat stress is low. For example, modelling of spring barley in central and western Europe showed that bringing the planting date forward by up to 60 days could lead to a 15-22% increase in yields under a doubled CO_2 climate, while using a long season cultivar could increase yield by 1.5% for each extra day of the growing season (Trnka *et al.*, 2004). Similar results were found by

Moriondo *et al.* (2010) for northern regions of Europe. If the planting date was delayed beyond the current one, then yields decrease (by 9.5% for a one month delay) due to higher temperatures and water stress. However, simulations of climate change and of planting maize two weeks earlier led to a 13% decline in yields (Tubiello *et al.*, 2000). Early planting of short season cultivars helps avoid summer heat and/or water stress (Tubiello *et al.*, 2000; Olesen & Bindi, 2002). Moriondo *et al.* (2010) found that on average the use of shorter cycle cultivars could decrease drought stress during the reproductive phase in southern regions of Europe by 12% and heat stress by 14%, while in northern regions the shorter growing season could reduce yields e.g. by 36% for sunflowers. However, in areas of high temperatures in the Mediterranean Basin it could lead to greater demands for irrigation for both types of cultivars (Giannakopoulos, 2009). The choice of cultivar is important if yield losses are to be avoided and potential gains from climate change realised, as was shown for soybeans in north east Austria (Alexandrov *et al.*, 2002) and maize in Romania (Cuculeanu *et al.*, 1999).

Winter crops

An investigation of rain fed crop production in Europe found that climate change could lead to an increase in the suitable days for sowing in autumn, although there could be higher variability in the conditions that limited sowing with consequential effects for yields (Trnka *et al.*, 2010; Peltoninen-Sainio et al, 2011). Studies have varied in their identification of best timing for sowing with modelling of winter wheat on the Great Plains, US, showing that the earliest date always did best (Southworth *et al.*, 2002), but that the possible later planting of spring crops of maize and soybean under climate change could lead to conflicts between harvesting them and the earlier planting time needed for winter crops to obtain maximum yields. While it has been suggested that in northern Europe, climate change may mean that sowing will have to be later in order to ensure that it occurs close to the time when vernalisation is most effective (Olesen *et al.*, 2000; Olesen *et al.*, 2004). Elsewhere cultivars better adapted to a warmer climate and requiring less vernalisation and with longer grain filling periods could be used (Tubiello *et al.*, 2002). In a study in north east Austria, Alexandrov *et al.* (2002) found that using winter wheat cultivars with a shorter vegetative growth period could increase yields under climate change. A comparison of winter and spring sowing of 19 to 23 ascochyta blight-resistant and cold-tolerant breeding lines of chickpea at three locations in Syria and Lebanon showed that seed yields of winter-sown chickpea were up to 70% higher and thus winter-sowing is increasingly being adopted for this crop (Singh *et al.*, 1997).

One of the adaptation strategies for wheat, maize and potatoes in the Great Plains involved the simulation of cultivars better adapted to a warmer climate, requiring less vernalization, and with longer grain filling periods (Tubiello *et al.*, 2002). The adaptation of cultivar was not seen as particularly necessary in order to maintain yields of wheat and maize given the projected climate changes, while potato production is mostly limited by the need for cold conditions for tuber initiation.

Breeding

Breeding can contribute to climate change adaptation through improving productivity, increasing drought-resistance (Dennis et al., 2000; Parry *et al.*, 2005; Smith *et al.*, 2012) and improving heat tolerance in livestock (Jordan, 2003; Nienaber and Hahn, 2007; Nardone *et al.* 2010). It can also contribute to mitigation through improved feed and reproductive efficiency, as well as improved growth rate. There are three main traditional strategies which have been used for the genetic improvement of crops and livestock: selection between breeds or strains, selection within breeds or strains, and cross-breeding. More recently, developments in genetic techniques, including gene transfer have been applied (Habash *et al.*, 2009).

Significant increases in crop yields have been recorded over the last 100 years as a consequence of breeding. A six year trial (2002 to 2008) of a historical set of 47 varieties of spring wheat (Triticum aestivum L.) developed and grown in Western Siberia between 1900 and 2000 (Morgounov *et al.*, 2010), analysed the genetic gains for grain yield and associated changes in agronomic traits for three maturity groups (early, medium and late) and four breeding periods (before 1930, 1950–1975, 1976–1985 and after 1985). It showed that the overall yield for modern varieties was 3.71 t ha^{-1} versus 2.18 t ha^{-1} for old varieties, which represents a 0.7% per annum increase over 100 years, although these figures are below the genetic gains reported in high-yielding environments for wheat. The grain yield difference between the newest varieties bred after 1985 and old varieties bred before 1930 for the early maturity group was 1612 kg (85%); for the medium maturity group, 1390 kg (58%); for the late maturity group, 1460 kg (62%), but the late maturity group showed a sharp decline in genetic gain with time, as has yield potential growth in favourable years. This suggests that conventional breeding may need to be supplemented by genetic interventions. A study of Canadian Western Red Spring Wheat class also showed that genetic gains in yield in 1984-2001 were much higher than in 1908-1986, due to more intensive breeding programs and increase of population size (de Pauw *et al.* 2007). This is mirrored in the UK where it was estimated that since 1982 around 90% of all yield increases in wheat and barley have been due to the introduction of new varieties and that fertilisers, pesticides and machinery have played a minor part, while remaining an important part of crop production (BSPB 2010). In Finland too, spring wheat between 1970-2005 has demonstrated consistent genetic gains of 36 kg y^{-1} (Peltonen-Sainio *et al.*, 2009). It has however been suggested that progress in yield attributed to breeding is inversely proportional to the stress in the growing environment (Richards, 1996), and in stressed environments the gains in wheat yield from modern cultivars has been less than needed to maintain food supply and is similar for barley (Araus, 2002). It has been estimated that the global average wheat yield will have to increase over the next 25 years from 2.6 to 3.5 tonnes per ha, and this will require a continuing supply of improved germplasm and appropriate agronomy (Ortiz *et al.*, 2008).

In organic agriculture (OA), however, there is a greater need for increased sustainability of performance of cereal varieties and more varieties adapted to organic conditions (Wolfe *et al.*, 2008). They suggest that even in Europe where OA is well-established, little specific breeding for OA has been undertaken. Also they suggest that breeding for conventional agriculture or centralised breeding will not meet the needs of OA, as more regional and local varieties are needed to cope with the greater environmental heterogeneity in OA.

Breeding is also being used to enhance crop efficiency of use of soil resources and coping with water (Ceccarelli *et al.*, 2007) and nutrient limitations (Lammerts van Bueren *et al.*, 2008; Sylvester-Bradley and Kindred, 2009). For example, Singh and Reddy (2011), using chickpea, identified genotypes and physiological parameters that could be used by breeding programs and/or genetic engineering for drought adaptation of legumes through increasing water use efficiency. It has been suggested that breeding for tolerance to low nitrogen may be related to tolerance to other stress factors and vice versa. Also, that drought and salinity are two of the most complex stress tolerances to breed for as the type and timing in relation to plant growth stage and intensity of stress can all vary considerably (Witcombe *et al.*, 2008). In addition, the traits associated with avoidance and tolerance can be constitutive (differing between genotypes) or adaptive (vary with the stage of the life cycle) and they involve different mechanisms and processes, with phenology as the single most important factor influencing whether a plant avoids drought (Witcombe *et al.* 2008).

Manderscheid and Weigel (1995) suggest that increases in CO_2, which can enhance long-term net assimilation and water-use efficiency, are responsible for nearly half the increase in yield of current cultivars under present day conditions, independent of a doubling of the harvest index[1]. They also concluded that, under good management, barley yield could increase by 0.35% per ppm increase in CO_2, whilst that for wheat would be about 25 % lower and both would be less in stressed environments.

Crop breeding and genetic modification in breeding to increase the productivity of bioenergy crops is another major research area (Ragauskas *et al.*, 2006; Groover, 2007). For example, the identification of the genes responsible for traits relating to increasing carbon partitioning to above-ground woody matter and increasing cellulose availability for enzymatic digestion (Groover, 2007). For animals, breeding is also important - not just directly for productivity, but also indirectly, for reducing heat stress which can adversely affect production. It is necessary to highlight here that measures such as this often involve trade-offs (see Section 10).

Water and irrigation

A number of papers identified a range of possible adaptation measures related to agricultural water use (e.g. Moriondo *et al.*, 2010) and many use modelling to explore possible future options, including what could happen if a particular measure was implemented under different climate scenarios (e.g. Rosenzweig *et al.*, 2004). Many of the papers examined, however, did not provide evidence of the use and effectiveness of current adaptation measures. This may be because farmers are constantly adapting to changing conditions and thus many potential impacts are theoretical (Reidsma et al., 2009). Also, as adaptation is a continuous process of intervention in various ways at different times, or in combination, the consequences of a particular measure cannot be easily assessed.

Tompkins *et al.* (2010), for example, commented that even in the UK there are relatively few agricultural examples of climate change adaptation, although many actions that could be considered adaptation (e.g. on-farm reservoirs) have occurred in response to legislation or other pressures, rather than directly for climate change. Some potentially relevant papers were in conference proceedings or specialised journals and were not available. Also, given that water is a bigger issue outside Europe, many papers were on developing countries and the tropics and were excluded from this review.

In most countries of Europe, agriculture is the major user of water, with irrigation taking about 70% of total available water (OECD, 2010), although in the Mediterranean, irrigation can account for about 90% of water consumption (Gómez-Limón and Riesgo, 2004). The need to manage water resources in the light of climate change, either through changing demand or providing/increasing supply through irrigation, especially in arid and semi-arid areas, is acknowledged by many (Falloon & Betts, 2010).

In an analysis of the implications of changing crop water demand and availability under different projections of climate change, agricultural production, population, technology, and GDP growth for selected countries (Argentina, Brazil, China, Hungary, Romania, and the

[1]Harvest Index – the weight of a harvested product as a percentage of the total plant weight of a crop.

US), Rosenzweig *et al.* (2004) show that only in the Brazilian case were there opportunities for increasing the irrigated area. In contrast, although Romania has potential for considerable expansion of its irrigated agriculture, both European countries in the study could suffer decreases in system reliability of upto 18%, as water stress is expected in some regions (southwestern and eastern) and in some model scenarios by 2020. In China, improved technology could lead to a decrease of about 20% in both water demand and demand-to-runoff ratio under a high efficiency irrigation scenario and even greater gains were shown for the Lower Missouri, US, which was sensitive to these technological improvements.

Modelling of the effect of global increases in agriculture and forestry on potential land and water use to 2030 led to projections of a need to expand irrigated areas by 14%, and consumptive irrigation water use by 7% when considering efficiency shifts (adaptation) based on irrigation method alone without technical innovation in agriculture (Sauer *et al.*, 2010). The highest absolute increase in irrigated area was projected for South Asia, while the highest relative increases of irrigation area expansion were found in the former Soviet Union, central and eastern Europe, North America and Latin America and the Caribbean. Globally, a general trend of combined expansion and extensification of irrigated agriculture was identified, with improved water use efficiency being driven by increasing rates of population growth. Any expansion of irrigated area, would, however, have implications for the wider water sector.

Modelling the impact of a 2°C increase in temperature on European agriculture showed that irrigation as an adaptation option was more beneficial in southern Europe (Moriondo *et al.*, 2010). In the Mediterranean, yields of sunflower, soybean, and spring wheat increased, on average, by 100%, 35% and 41%, respectively, in response to an irrigation of 142, 70, and 120 mm ha^{-1} $season^{-1}$, respectively, while in northern Europe, yields of sunflower, soybean and spring wheat increased by 60%, 27%, and 15% with the application of 76, 40 and 35 mm ha^{-1} $season^{-1}$ of water, respectively. In Spain, agriculture accounts for about 80% of water consumption and the importance of irrigation for maintaining yield is exemplified by modelling in the Ebro Basin of a reduction of irrigated area by 10, 20 and 30% (Gómez-Limón and Riesgo, 2004). It showed that this could lead to a decrease in yield ranging from 2% for wheat to up to 15.5% for alfalfa depending on the scenario.

A widely practised alternative approach is deficit irrigation, where crops are deliberately under-irrigated and crops stressed with the intention of affecting yield, economic returns or water usage (Mushtaq and Moghaddasi, 2011). A number of studies have shown that water application can be reduced without a significant decrease in yields (e.g. Kirda *et al.*, 1999; Gorantiwar and Smout, 2003). Mushtaq and Moghaddasi (2011) used scenarios to explore the effect of different irrigation strategies (optimization with full irrigation, optimization with deficit irrigation and deficit irrigation without optimization) on crop production and profits. They found that deficit irrigation (equally reducing water use for each crop to calculate the impact of gross margins on total gross margins) led to decreases in yield of 57% for pasture and 39% for wheat, but crops showed different sensitivities to the reductions and thus a more targeted approach towards sensitive crops was suggested. The best scenario was optimization with deficit irrigation.

Adaptation through irrigation water management efficiency and cropping patterns may not be sufficient to prevent an increase in water requirements (Purkey *et al.*, 2008). Modelling a combination of the two adaptation measures for the Sacromento Valley, US, showed that they could reduce future demand to close to current levels, but the effectiveness of each depended partly on the type and strength of water rights in the different districts in the basin. Also, while the reduced demand freed up water for other users, it led to little overall change in demand and they advocated an integrated approach to water management (Purkey *et al.*, 2008). An alternative approach to water scarcity would be for producers to reduce activities which require irriation. Forexample, in the Murray-Darling Basin, Quiggin *et al.* (2010) using a state-contingent model showed a possible shifting of horticultural production from citrus and grapes to vegetables or rock melons which do not require irrigation even under drought conditions. A reduction in irrigated area, and altered cropping patterns have already been observed here since the start of the drought in 2001/2 (Sanders *et al.*, 2010).

The adoption of specific crop management options (e.g. changes in sowing dates or cultivars) also may help in reducing the negative responses of agricultural crops to climate change. However, such options could require up to 40% more water for irrigation, which may or may not be available in the future (Giannakopoulos et al., 2004).

It may not just be a matter of adapting to reduced water supply, but also, particularly in arid and semi-arid regions, to increases in its variability and salinity. A good example is from the Murray-Darling Basin, where research showed that if these two additional factors are not taken into account then the impacts of climate change will be underestimated (Connor *et al.*, 2012). Possible adaptations identified to cope with variability included decreased planting of perennial crops, while salinity could be addressed by increased application of water to leach the salt, but this needed to be balanced by greater fallow elsewhere.

Climate change is only one of a number of variables that will affect European agriculture. In an analysis of the impacts of changes in climate, subsidies and farm inputs and outputs, showed that irrigation only produces a small increase in output, although it is shown to be a good adaptation option to climate change in Greece (Reidsma et al., 2009).

There is less evidence of adaptation in pastoral farming, but, in northern Victoria, Australia, the drought which began in 2001/2002 has forced farmers who used to flood irrigate their perennial pastures in the summer months to switch to using forage species, such as maize, annual ryegrass and lucerne, which are more water-efficient during these drier months (Henry *et al.*, 2012). This has increased water use efficiency, as well as increasing total annual production, although other factors such as nutritive characteristics, cost of production, and cost of transferring feed need to be considered when choosing what to grow (Lawson *et al.* 2009). For livestock, adequate irrigation water availability may be critical in enabling projected increases in growth in annual pasture systems to be realised (Cullen and Eckhard, 2011), although they still use less water than perennial irrigated pastures.

High(er) temperatures, such as will be experienced under climate change, can negatively affect the physiology and productivity of cattle (West, 2003) and one adaptation is the provision of shade to reduce heat stress. Cattle have been shown to seek shade as temperatures rise to 28°C (Fraser & Broom, 1997) or 30°C (Titto et al., 2011). The shade can be artificial (e.g. roofs, shelters) or natural (trees). Studies have shown the latter to be preferentially chosen by beef cattle (Shearer *et al.*, 1991), but Gaughan *et al.*, (1998) found that Holstein-Friesian cows preferred iron roofs to trees. Trees have been found to be more effective in reducing temperatures (by up to 2°C; Bray *et al.*, 1994) and in increasing productivity through longer times spent grazing (Titto et al., 2011). Overuse of shade trees, however, can lead to their mortality as roots become exposed and soil oxygen levels decrease due to compaction. This can partly be overcome through providing more trees and moving stock between shaded areas.

Soil management practices

There were a relatively small number of articles discussing soil management practices specifically as a form of climate change adaptation, with many failing to identify a direct link, noting instead benefits to farm productivity, or improvements in water quality. A number of articles for locations outside of Europe did, however, explore the potential of no-, or reduced-tillage practices as a form of adaptation to climate change, whereas few examples existed for Europe, these being located in areas with semi-arid climates, such as the Mediterranean Basin (Kassam *et al.*, 2012).

The increased water holding capacity of the soils (Klik & Eitzinger, 2010) was a key factor in the potential for conservation agriculture practices to be employed as a form of climate change adaptation, with practices such as direct seeding shown to conserve soil moisture content in dry regions such as parts of the Mediterranean as a result of cover from residues etc. (Munoz *et al.*, 2007). As a result, these agricultural systems are less vulnerable to drought conditions, which are expected to increase in severity as a result of climate change (Kassam *et al.*, 2012). In addition, Carlton *et al.* (2012) suggest that reduced tillage practices may be important in the future in southern and eastern areas of England, again for the same reason as above. A study by Desjardins *et al.* (2005) found conservation soils to have a higher soil moisture content as a result of the organic residue cover in these agricultural systems, and therefore suggests that this may allow for year-round cropping in semi-arid agricultural zones, resulting in reduced summer ploughing, in addition to increases in long-term crop production and carbon inputs to the soil surface in some locations. In addition, Oorts *et al.* (2007) identified no-tillage plots to have a lower soil temperature during the summer than those under conventional tillage methods.

As with much of the mitigation studies for conservation agriculture, the results from this search have highlighted that the apparent ability of this practice to increase resilience to drought is, however, not consistent throughout the literature. For example, studies have found that crop water use efficiency does not always increase under conservation tillage management practices (e.g. Cantero-Martinez *et al.*, 2007; de Vita *et al.*, 2007). In addition, a study by van den Putte *et al.* (2010) found no-till practices to perform worse under a drier climate, as a result of secondary effects such as an increased abundance of pests and a lower quality of seed placement.

3.1.2 Agriculture in China

In China, major climate-related stresses on agriculture include regional temperature distribution, change of rainfall frequency and severity, CO_2 concentration enrichment and sea level rise. These could lead to a series of impacts, such as warmer and drier environments, increased frequency and severity of droughts in the north, increased frequency and severity of floods in the south, and extreme temperatures, etc. (Table 2). Corresponding to those impacts, there are a lot of observed and projected adaptation options, which could be categorized into four groups (Table 3), i.e. structural measures, agricultural practices, technological change and management and policies. Many of these are similar to adaptation practices in Europe. However, present adaptation research in China is supplementary to impact research and there is no systematic review of China's adaptation for agriculture. Generally speaking, China's adaptation for agriculture is leaping from traditional agricultural practices, such as water-saving irrigation, terracing of sloping land, water storage, etc., to being more dynamic in terms of infrastructure, new technology, macro-management and on-field management practices to maximum cost-benefits (Deng *et al.*, 2010; Zhou *et al.*, 2010). On the basis of the existing literature review, these were the eight measures with the most hits.

Table 2: Impacts of climate change on agriculture and adaptation measures in China.

Climate-related change/ stress	Impacts	Adaptation practices
Temperature	Warmer and drying environment	Irrigation Improved water management Use of different species better adapted to the warmer and drier environment
	Increased frequency and severity of droughts in the north	Breeding for flood tolerance Fitting the pattern of crop growth and development to the availability of soil water Breeding and selection for yield in water-scarce environments
	Northward movement of crop suitability zones	Adjustment of crop patterns, e.g. the expansion of rice area in northeast China
	Earlier planting of crops	Longer-season cultivars
	SOC decomposition acceleration	Fertilizer management
Rainfall frequency and severity	Increased frequency and severity of floods in the south	Structural measures (reservoirs)
	Drying environment and water scarcity	Structural measures (reservoirs) Water saving agriculture Irrigation
CO_2 concentration	Pest diffusion	Integrated pest management
Sea level	Threats from sea level increase in the coastal area	Structural measures (dykes, dams, etc.) Flood prevention standards

Sources: Du *et al.* (2009); Wang & Ma (2009); Zhou *et al.* (2010); Pan *et al.* (2011); Jin & Gao (2012).

Table 3: Classification of existing adaptation practices in agriculture in China.

Types	Adaptation practices
Structural measures	Water and irrigation infrastructure, such as reservoirs, drilling wells, drainage systems, water storage facility, water supply system etc. Tidal or river flood prevention infrastructure, such as dams.
	Intra-basin water transfer projects
Agricultural practices	Water-saving irrigation
	Varieties of crop planted, better variety adjusted to the warmer and drier environment
	Planting time adjustment
	Multiple cropping
	Conservation/no tillage
	Weed and pest control
	Terracing of sloping land
	Water storage
	Mulching (plastic sheet)
Technological change	Breeding selection (long-season cultivar, breeding for heat tolerance, etc.)
	Genetic modified organisms (GMOs)
Management and policies	Land planning and management
	Disaster early-warning system
	Fertilizer management
	Flood prevention standards
	Integrated coastal management
	Agricultural insurance

Breeding

Global temperature increase is expected to accelerate the growth of all crops. As observed by Cooper *et al.* (2009) and Vadez *et al.* (2011), higher temperatures speed up flowering and maturity and then shorten the time from sowing to maturity. Contrary to the current practice, warmer temperatures can lead to water remaining in the soil profile in medium rainfall areas (290 mm growing-season rainfall) after harvest, as plants have a lower leaf area and biomass.

This suggests that, as climate warms in semi-arid environments, breeders should select for longer-season cultivars so that crops can take advantage of this water (Turner *et al.*, 2011). However, when the growing-season rainfall was 180 mm, longer-season cultivars had no yield benefit (Turner *et al.*, 2011). In most parts of the Loess Plateau where soils are deep, rainfall is above 200 mm and predicted to increase, and crops grow on a mixture of current rainfall and stored soil moisture, Turner *et al.* (2011) suggested that longer-season cultivars would be particularly beneficial. In the Songnen Plain, longer-season cultivars of maize and wheat have been selected to adapt to earlier planting (Zhou *et al.*, 2010; Xie *et al.*, 2011). In Shandong and Gansu province, on the basis of results from CERES modelling and scenario evaluation, longer-season cultivars led to increased crop yields of wheat and cotton (Yuan and Xu, 2008; Chen *et al.*, 2011). In the middle and downstream area of the Yangtze River, new species better adapted to climate change are regarded as a very effective way to stabilize crop yields. Model results also showed that new soybean species with better heat tolerance could increase crop yields by 13-22% and 4-15% respectively in 2030 and 2050 (Shi *et al.*, 2001; Ge *et al.*, 2002).

Breeding for heat tolerance is needed among the major crops grown in the Loess Plateau, South China, Middle China and East China (Zhou *et al.*, 2010; Turner *et al.*, 2011). Breeding for flood tolerance is also needed in South China, Middle China and East China, since extreme rainfall there is projected to increase (Zhou *et al.*, 2010). However, Tao and Zhang (2010) suggest that for some high-temperature sensitive varieties early planting should be a generally effective adaptation option to reduce yield loss from climate change, while for some high-temperature tolerant varieties late planting could be a generally effective adaptation option.

Yields have been increased in drought-prone environments by decreasing the time to flowering and maturity so that crops avoid terminal drought induced by a lack of rainfall or by premature use of stored soil moisture, and by fitting the pattern of crop growth and development to the availability of soil water (Siddique *et al.*, 1990, 2001; Turner *et al.*, 2001; Turner, 2004a). Breeding and selection for yield in water-scarce environments has traditionally been employed, but more recently physiological attributes for improved drought resistance have been sought and evaluated (Turner and Asseng, 2005; Richards, 2006). To adapt to a drier climate, Liaoning Academy of Agricultural Sciences also selected some genotypes of upland rice with higher water use efficiency in northeast China and now the cultivated area of upland rice has increased to more than 20, 000 ha in the past decade (Xie *et al.* 2011).

<u>Irrigation</u>

Irrigation is one important strategy to defend against and mitigate drought and it is also important for improving crop yields, protecting water supplies, ensuring food security, increasing income and improving the ecological environment (Wu *et al.*, 2011). Strengthening irrigation capacity is regarded as one of the most beneficial means to maintain agricultural production in the face of unfavourable climate change (Lin, 1996). However, most irrigation infrastructure in rural areas in China was constructed in the 1950s and could not meet the demands of a changing climate (Zhou *et al.*, 2010). China has proposed an adaptation strategy of increasing food production through irrigated agriculture (You, 2001).

In the coming decades, China will have to face insufficient water for agriculture due to a warmer and drying climate. However, as projected by Xiong *et al.* (2010), the shortfall in irrigation area is estimated at 27 Mha and 15 Mha respectively for the A2 and B2 socio-economic development pathways. Water-saving irrigation can help reduce the negative impact of climate change on water resources available to agriculture and overcome the constraint of water scarcity by reducing water consumption and increasing water productivity (Belder *et al.*, 2005; Tuong *et al.*, 2005). The extent to which reductions could be achieved is shown in Table 4. In this case, Chinese basic national policy has highlighted water-saving irrigation as an important component for boosting sustainable agriculture, as well as the coping capacity of agriculture for climate change. A host of field-level experiments has reported that water saving irrigation could contribute to climate change adaptation by reducing water consumption, increasing water use efficiency and increasing crop yields in northern China (Liu *et al.*, 2003; Deng *et al.*, 2006; Du *et al.*, 2010; Yang *et al.*, 2010). In the North China Plain, Liu *et al.* (2007) found that irrigation water demand could be reduced by 5-25% by reducing irrigation depth and Zhang *et al.* (2006) found that improved soil water condition could offset the negative impacts on crop yields by 5.2%. In southern China, irrigated rice will increase the yields by 1-2 times compared with rainfed rice, which will offset the negative impacts of climate change over the next 50 years (Ge *et al.*, 2002).

Table 4: Water saving potential for major irrigation technologies in China.

Irrigation technology	Water saving potential*
Drip irrigation	30-40%
Sprinkler irrigation	40-50%
Small furrow, pipe	20-25%
Subsurface irrigation	20-25%
Surge, intermittent	15-20%
Low pressure hose	30%

Source: Liu and Li (2002).
*Water saving potential is calculated on the basis of water consumption of surface flooding irrigation.

Tillage

Minimum tillage is not widely used on the Loess Plateau of northwest China, but in rural areas where two- or three-wheeled farm tractors and power tillers are widespread conversion of power tillers for minimum tillage is now being adopted (Siddique *et al.*, 2001). However, the technique is beginning to be evaluated in China (Liu *et al.*, 2009; Yang *et al.* 2010). Since 2000, conservation tillage characterized by crop returning and less tillage has been experimented with and extended in northeast China (Xie *et al.* 2011). Field experiments showed that conservation tillage could increase the water storage ability and water content of soil up to a depth of 200 cm. Compared to traditional tillage, no-tillage and cover crops could increase the soil water content by 1.93-7.25% and 0.06-3.58% respectively, and cover crops can increase the water use efficiency by 30-40% (Guo *et al.*, 2005). Similar results were found by Li *et al.* (2002). For this reason, some researchers believe that conservation tillage could help to tackle water scarcity due to drying climate (Liu *et al.*, 2006). However, other researchers argue that although results show that conservation tillage has an ability to reduce soil erosion, increase soil organic content, reduce the water demand of crops and enhance crop productivity, they have not provided marked evidence for benefiting climate change adaptation, particularly in cold provinces in China where the mineralisation of crop residues is slow (Xie *et al.*, 2011).

Rotation and fallowing

Terracing of sloping land has been widely adopted to reduce soil erosion and runoff and to conserve water for crop production (Cao *et al.*, 2007; Gao and Deng, 2007), particularly for high value crops such as apples and other fruits. Plastic mulching is not only used to warm the soil for earlier planting, but also to reduce soil evaporation, focus precipitation in the root zone and conserve soil water in the fallow season (Tian *et al.*, 2003; Liu *et al.*, 2009; Zhou *et al.*, 2009). Water catchment and storage is utilised on the Loess Plateau to provide supplementary irrigation, which has been shown to increase yield and water use efficiency of crops in semi-arid regions (Li *et al.*, 2001; 2004).

The purpose of fallowing is to conserve water from one season to another. Numerous authors have studied the efficacy of fallowing (duration, management of crop residue, tillage, etc.) for storing water for the subsequent crop. The efficacy of fallowing as regards to the transpiration of the succeeding crop may be extremely variable depending on soil depth, texture and structure and whether weeds are controlled (McAneney and Arrue, 1993).

Weed and pest control

Weed and pest control are considered important in conserving water and maximising yields as water becomes scarce and increased temperatures favour weed and pest development (Turner, 2004a, 2004b, 2011; Turner and Asseng, 2005). Development of integrated pest management, use of genetically-modified crops with insect- and herbicide- resistant genes, and the use of rotation to control weeds and pests have become important requirements as climate changes in northwest China.

Planting time adjustment

Earlier planting of photo-insensitive crops, particularly using longer-season cultivars developed through breeding, helps adaptation to minimum temperature and a decrease in frost risk with the warming climate; earlier planting and earlier flowering provide a better fit between the growth pattern and soil water availability (Siddique *et al.*, 1990; Turner 2004a). On the Loess Plateau, where crops usually grow over summer and frost risk at flowering is not an issue, plastic film mulch has been introduced to warm the soil in spring and allow earlier planting of spring wheat and maize after winter (Li *et al.*, 1999, 2004, 2009; Zhou *et al.*, 2009). In the middle and downstream areas of the Yangtze River catchment, postponing the planting time for 20 days can increase the yields of early rice by 4.8% and 9.1% respectively in 2020 and 2050 (Shi *et al.*, 2001).

However, it is very difficult to judge the benefit of earlier or later planting time, since simulation in Shandong and Gansu province showed that the time of planting led to different impacts on productivity under different climate scenarios (Jin *et al.*, 1998; Yuan and Xu, 2008; Chen *et al.*, 2011). In this context, Deng *et al.* (2010) suggested adjusting the time of planting according to climate pattern.

Adjustment of crop varieties

On the Loess Plateau, the use of the perennial fodder crop lucerne is being adopted to supplement maize and wheat straw for penned animals (Wen *et al.*, 2003) and identification of fodder species better adapted to warmer temperature, such as fodder sorghum is warranted. Similar results were found by Yuan and Xu (2008) for wheat in Shandong province. However, although screening suitable crop variety is regarded as a very effective way to reduce climate change, only medium-sized farms would take such measures (Chen *et al.*, 2010).

Early-warning and risk management system

At seasonal or yearly timescales, early warning and risk management systems are obviously an efficient way to reduce disaster and can facilitate adaptation to climate variability and change (Meza and Wilks, 2003; Hansen *et al.*, 2006). Those adaptation options can particularly be applicable in the North China Plain, where climate variability associated with the East Asia summer monsoon and ENSO resulted in considerable yield variability (Tao *et al.*, 2004). In the south, professional officers in the County Bureau of Plant Protection in Jianghan Plain will deliver some guidelines in the form of newspapers on climate risk, which would tell the farmers how to prevent the risks (Chen *et al.*, 2010).

3.2 Biodiversity

Adaptation for biodiversity has been strongly promoted as a result of various international targets, such as the Convention on Biological Diversity's 2010 target, which was followed by the Aichi targets. The latter include Target 11 which states that "By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes."[2]. The EU, as signatories to the CBD undertook to halt the decline of biodiversity in the EU by 2010 and to restore habitats and natural systems, as with the CBD 2010 target it was not fully met and so they have adopted a 2020 strategy "to halt the loss of biodiversity and ecosystem services in the EU by 2020".[3] Many of the actions required to meet these targets, such as corridors to improve ecological connectivity and species movements and improvement of condition of habitats and protected areas, are also consistent with the adaptation of biodiversity to climate change and it is difficult to separate which have been undertaken specifically for climate change and which are part of a drive to reduce biodiversity loss and habitat degradation (Target 14 "By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded…"). Nevertheless, the EU sees the two issues of biodiversity loss and climate change as inextricably linked[4]. The five measures that are most widely promoted in the literature are discussed below.

3.2.1 Assisted colonization

Assisted colonization (also termed assisted migration, managed relocation, or translocation) is a possible method of adaptation, and a way to reduce the risk of extinction for species with low dispersal rates, or patchy habitat distributions (Kreyling *et al.*, 2011; Thomas, 2011).

[2] http://www.cbd.int/sp/targets/

[3] http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/1_EN_ACT_part1_v7%5B1%5D.pdf

[4] http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/1_EN_ACT_part1_v7%5B1%5D.pdf

This measure is therefore often seen as being particularly appropriate for the conservation and restoration of systems such as forests in response to climate change (e.g. Chapin *et al.*, 2007; McKenney *et al.*, 2009).

Research has highlighted the need for such a management option to be employed in Europe, with a number of species in southern Europe along the Iberian Peninsula classified as high risk due to a strong dispersal limitation (Araujo *et al.*, 2004; Svenning *et al.*, 2009; Thomas, 2011). One study which assesses the potential for assisted colonization as a management option in Europe is that of Morueta-Holme *et al.* (2010). The authors assessed changes in species distributions of a number of small mammal species for the period 2070-2099, finding significant reductions in species ranges, which could lead to the near total extinction of the Pyrenean Desman (*G.pyrenaicus*) in Spain. Assisted migration may be therefore be able to help certain species such as this with low dispersal capacity adapt to climate change, by moving them to suitable future climatic space (Morueta-Holme *et al.*, 2010).

As far as flora are concerned, it has been calculated that there exists considerable space for additional plant species, with half of northern Europe thought to be capable of hosting over a third as many new additional species as they currently have native species (Svenning *et al.*, 2009). Hence, in theory it is possible to conduct assisted migration as a conservation strategy in response to climate change, without this adversely affecting native flora in selected target areas (Svenning *et al.*, 2009).

Numerous recipient regions, usually in northern Europe, for translocated species have been considered, with Britain often identified as an ideal recipient location (e.g. Carroll *et al.*, 2009; Thomas, 2011). Thomas (2011) for example identified the UK as a suitable site for the translocation of a number of highly endangered species, including the Iberian lynx (*Lynx pardinus*) and Spanish imperial eagle (*Aquila heliacea adalberti*). Suitable prey already exists for these species in the UK, and it is predicted the area will become suitable in terms of climate space for these species as a result of climate change. These translocations could be key, as it is thought the establishment of the Iberian lynx in the UK would contribute more to biodiversity conservation efforts than the re-introduction of the Eurasian lynx (Thomas, 2011).

The literature review found only one example of the assisted migration in practice, with the two butterfly species; *Melanargia galathea* and *Thymelicus* sylvestris introduced to the UK (Willis *et al.*, 2009). Areas were identified by climate models as being suitable for the species; containing both suitable habitat area, and future climatic space. These species were then introduced to areas of northern England, with *Melanargia galathea* relocated 65 km beyond the range of its current distribution, and *Thymelicus sylvestris* 35 km beyond its current range. Despite an initial colonization lag, these relocations were successful, with both species populations having expanded their range after translocation, and continuing to bloom many generations later (Willis *et al.*, 2009). The success of this study demonstrates how assisted colonization can be a viable adaptation option for species of conservation priority with patchy habitat distributions and poor dispersal capability.

Despite the apparent limited uptake of assisted colonisation programmes in Europe, the literature search found evidence of this adaptation policy being undertaken in other countries such as North America. One example here is that of a volunteer organisation managing the assisted colonization of *Torreya taxifolia*, a conifer from the south-eastern US, whose native range is declining substantially, most likely as a result of climate change (Barlow & Martin, 2007).

Perhaps the restricted number of studies discussing cases of managed relocation for the purpose of climate change adaptation is a result of the many unknowns, concerns over technical feasibility, and potential for secondary effects (Hunter, 2007; Mueller and Hellmann, 2008; Pelini *et al.*, 2009). For example, although unlikely, if a translocated species were to become invasive, this could have substantial adverse impacts on native species (Mueller and Hellmann, 2008). In addition, the introduction of a non-native species can lead to the spread of disease and pests (Ricciardi and Simberloff, 2009), and severely impact both the functioning and composition of an ecosystem, with numerous instances of extinctions documented following past species introductions (Etterson, 2008; Hoegh-Guldberg *et al.*, 2008). These potential adverse secondary-effects are numerous and result in the chosen recipient location being at high risk (Davidson and Simkanin, 2008; Mueller and Hellmann, 2008; Ricciardi and Simberloff, 2009). As a result of this, some authors find assisted colonization to be either an infeasible conservation strategy, or one with limited potential (Sandler, 2010).

In contrast, and despite these uncertainties, a review by Ricciardi and Simberloff (2009) found the risks associated with assisted migration to be fairly low, especially for regional or intra-continental translocations (Ricciardi and Simberloff, 2009; see also Morueta-Holme *et al.*, 2010). What is more, the risks resulting from a failure to act are thought to be much higher than those associated with the adoption of this management strategy (Schwartz *et al.*, 2009). Finally, further minimisation of the risks surrounding translocations can be achieved by examining the effects of numerous historical species introductions, as well as other factors (Hoegh-Guldberg *et al.*, 2008; Loss *et al.*, 2011; Mueller and Hellmann, 2008; Willis *et al.*, 2009).

3.2.2 Corridors

Corridors provide a pathway for species between protected areas (PAs), and hence improve the connectivity of ecosystems, and the ability of species to migrate (e.g. Pearson and Dawson, 2005). Temporal corridors, representing an overlap between existing and projected future corridors have also been discussed in the literature (Rose and Burton, 2009). This literature search found no specific examples of such corridors being used in Europe, although it is noted that these would give priority to the siting of new protected areas in such locations (Rose and Burton, 2009).

The creation of corridors across Europe is seen as key for conserving target populations such as those of the yellow-legged dragonfly (*Gomphus flavipes*), a highly vulnerable species included in the European and IUCN Red-Lists (van der Sluis *et al.*, 2004). In the past, water pollution and loss of habitat restricted these species to locations along the Loire, and rivers such as the Elbe and Spree in eastern Germany. However, in recent decades they have undergone a sudden recent expansion, appearing in both the Netherlands and areas of western Europe; most likely in response to the warming climate (van der Sluis *et al.*, 2004). The creation of corridors intended for *Gomphus flavipes* would create new habitat space, allowing the larvae, which favour warmer climates to migrate to suitable climatic space, whilst serving as stepping stones for flying adults (van der Sluis *et al.*, 2004).

Evidence was found of corridors being created in the Netherlands as part of the de Doorbraak project (WRD, 2011). This consists of a 13 km long new stream to increase species resilience to climate change by providing a corridor to other regions, such as northeast Tente and the Crest of Salland, protecting both fish and amphibian species (WRD, 2011). This scheme also has a number of synergies with the water sector (Section 7.2.2).

It is important, as far as support for this option is concerned, that it is generally favoured over assisted migration, seen to have lower risk, with no known instances of this having caused the spread of invasive species (Krosby *et al.*, 2010). Despite this, there remains the problem of timing, as natural dispersal may not occur at a sufficient rate for many species, hence some authors suggest that large-scale corridors may not be effective in all cases for helping all species (especially those with low dispersal rates) adapt to climate change (Pearson and Dawson, 2005).

3.2.3 Refugia

This study was unable to find evidence of refugia being either identified or protected in Europe as a form of climate change adaptation (see Keppel *et al.*, 2012).

3.2.4 Networks

The need for networks, linking corridors and PAs in the adaptation of biodiversity to climate change has been identified by a number of European countries. For example, in Germany it is predicted that as much as 30% of the country's current plant and animal species could become extinct in a time frame of decades as a result of climate change, with those in the Wadden Sea tidal flats being particularly vulnerable (BMU, 2008). Therefore, as part of the German Adaptation to Climate Change Strategy, and the National Strategy on Biological Diversity, the German Federal Government recognises that Länder should improve networks to allow species and populations to migrate northwards in response to climate change, and are therefore taking precautionary measures to aid adaptation (BMU, 2008). These networks, which could be included in schemes such as CAP, and the German National Water Meadows Programme, will need to cross borders and therefore their success is reliant on the collaborative efforts of actors across Länder and European country borders (BMU, 2008). Despite this potential, there exists a possible conflict between actors, with the land use requirements for networks being in competition with those from the agriculture and forestry sectors for example (BMU, 2008).

On a regional scale, the Natura 2000 Network[5] is one programme seeking to improve connectivity across Europe. Little was found in the literature search specifically linking the creation of this network for climate change adaptation, focussing mainly on restoration aims, although this existing network will aid migration by ensuring connectivity between present and future suitable climatic space (Natura 2000, 2007). Climate-change-proof assessments can be used to assess the resilience of such networks to climate change. These consist of a three-step process in which firstly existing habitat is mapped; secondly, current habitat networks are identified; and finally it is estimated how these networks may change as a result of warming (Vos *et al.*, 2008). A study by Araujo *et al.* (2011) found that compared to areas not covered by the network, Natura 2000 sites do not function significantly better for plant species under climate change. For example, loss of plant and animal species due to climate change was modelled to result in a 63 ± 2.1% loss of suitable area for species covered in the Bird and Habitat Directives in Natura 2000 sites, being in some cases less effective at conserving species than unprotected areas (Araujo *et al.*, 2011). This is partly due to much of the network being located on flat areas, where the effects of climate change will be greatest (Araujo *et al.*, 2011).

The European Green Belt (EGB) is another biodiversity network acting on a European scale found in the literature. This network operates across 24 countries, aiming to increase cross-

[5] http://www.natura.org/about.html Accessed 06/08/2012

border connectivity, thus aiding the dispersal of species across areas of Europe (Zmelik *et al.*, 2011). The EGB consists of a number of conservation areas including Ramsar sites, National Parks and parts of the Natura 2000 network, all with varying levels of protection. It is home to 243 species of the EU Habitats Directive, with rare and endangered species such as the Balkan Lynx and Dalmatian Pelican (Zmelik *et al.*, 2011). The green belt is rich in biodiversity, extending along the former Soviet border with habitats ranging from the boreal and tundra landscapes of Fennoscandia, to the high mountain areas in the Balkans, and the agricultural landscapes in central Europe. What is most significant about this corridor, or network, is its north-south gradient, which if managed appropriately should be capable of facilitating species migration whilst averting significant losses, hence it has been termed a 'climate change mitigating corridor' (Zmelik *et al.*, 2011).

Examining the extent to which broad networks across Europe are resilient to future change, Vos *et al.* (2008) conducted a climate-change-proof assessment of nine species in north-western Europe, examining three forest species (black woodpecker, middle spotted woodpecker, agile frog); three wetland species (bittern, marsh warbler, large heath butterfly); and three natural grassland species (brown hare, meadow pipit, and pool frog). The assessment reveals a future reduction in suitable area for all species surveyed, including a decline in suitable habitat provided by the Natura 2000 network (Vos *et al.*, 2008). It is important to note that the magnitude of this reduction is highly variable, for example with suitable area for the agile frog declining by only 6%, whereas the black woodpecker, marsh warbler and meadow pipit all could see declines in suitable area of 70% (Vos *et al.*, 2008). Taking the middle spotted woodpecker as an example, results show that although this species will lose suitable habitat space in regions such as France, it is projected to expand northwards into Ireland, the Netherlands, and Denmark, as well as the north of England and Scotland. Despite the climatic space in these locations being suitable for migration, these areas are too isolated to be successfully colonized by the species on either a 2020 or 2050 timescale, with bottlenecks predicted to occur (Vos *et al.*, 2008). As far as the proportion of the middle spotted woodpecker population in England is concerned, overlapping current and future distributions in 2020 in what are termed 'climate proof networks' reveals only a small overlap (less than 20%). Hence, the middle spotted woodpecker will have limited capacity to colonize new climate space in the UK unless adaptive measures, such as those to increase connectivity, and the integration of countryside management are taken (Vos *et al.*, 2008).

3.2.5 Habitat restoration

Restoration programmes for the purpose of climate change adaptation, or to increase resilience to climate change are scarce in the literature, being conducted mainly to restore the damage from human-induced stressors, rather than for a long-term future purpose such as climate change.

The Restoring Peatlands Project is, however, one example of restoration discussed as an adaptation option; aiming to restore substantial areas of degraded peatland in both Belarus and the Ukraine (see restoringpeatlands.org). This project will result in the provision of suitable habitat for a number of species, helping to conserve diversity. As far as synergies are concerned, this adaptation measure reduces GHG emissions, hence mitigating climate change, with the branch of the project in Belarus estimated to sequester a total of 2.9 tons CO_2 equivalent ha^{-1} y^{-1}. In addition, the restored peatlands regulate the local micro-climate, improve soil quality, can reduce the likelihood of peat fires, and impact positively on the water sector; improving water regulation and retention, as well as stabilizing the water level via a series of dams and reservoirs. The scale of the scheme varies between the two countries, with 14,000 ha peatland being rewetted in Belarus and 20,000 ha in the Ukraine at a total cost of €7.4 million (see Section 15.1.2 for more details on the economics of biodiversity projects). Both projects are being undertaken with support from a number of actors, for example the rewetting project in Belarus being coordinated by the RSPB, APB-Birdlife Belarus and the German Michael Succow Foundation.

3.2.6 Protected Areas

Species distribution models and gap analysis have been used to identify the most appropriate areas for environmental protection. This can lead to the siting of new PAs, either protecting future habitat space, or current habitat to increase resilience.

This literature search found a large number of studies discussing the ability of PA networks to aid species adaptation to climate change, and hence limit the number of species lost (see Hannah *et al.*, 2002; 2007, and references therein). More recently, transboundary conservation areas (TBCAs), where new PAs are constructed to enhance current PAs, as well as extending current management across borders, have also been considered as a way to facilitate species adaptation to changes in climatic range (Hannah, 2010).

The limited ability of existing PAs to aid species adaptation to climate change is stressed in the current literature, as they are expected to undergo changes in their functioning, and species composition as a result of climate change (Klausmeyer and Shaw, 2009; Carvalho *et al.*, 2011). Climatic zones such as the Mediterranean are expected to expand into northern areas for the period 2070-2099 (Klausmeyer and Shaw, 2009; Hannah, 2010), and therefore it has been calculated that the existing PA network in the Iberian Peninsula will need to be 1.15-1.89 times larger if it is to represent the same proportion of herptile species under a future climate (Carvalho *et al.*, 2011). Despite this apparent growth, species located in areas such as Iberia are expected to experience a substantial contraction in number, as there are no bordering areas in the Mediterranean basin for species to expand into (Hannah, 2010). For similar reasons, the climatic range of 1,200 European plant species is expected to contract by 6-11% over the next 50 years (Araujo *et al.*, 2004). PAs located in dry parts of the Mediterranean basin have been identified as those most susceptible to the effects of climate change, with Spain projected to experience a reduction of 43-29% in mammalian species richness depending on the climate scenario employed (Maiorano *et al.*, 2011). In contrast, PAs in regions of high altitudinal gradients such as the Apennine mountain range in Italy, as well as areas in France and northern Finland are expected to see increases in species richness as a result of climate change (Pöyry and Toivonen, 2005; Maiorano *et al.*, 2011).

New PAs would represent a larger number of species and could act as stepping stones, increasing connectivity between networks (Araujo *et al.*, 2004; Vos *et al.*, 2008; Hodgson *et al.*, 2011). As far as the location of suitable sites is considered, the framework of ‘conservation planning’ can be employed, in which new PAs are planned depending on the robustness of uncertainty in future species distributions (Carvalho *et al.*, 2011; see also Araujo *et al.*, 2004). For example, areas projected to be adversely affected with mild levels of uncertainty should be considered as sites for new conservation areas if funds permit, and subsequently those associated with very high levels of uncertainty should be given less priority. Interestingly, areas with the lowest levels of uncertainty are often already managed as PAs (Lawler, 2009; Carvalho *et al.*, 2011). This uncertainty may be one reason for the lack of specific examples of new PAs, and indeed other measures being undertaken to facilitate species migration in response to climate change.

In highly sensitive areas such as the Iberian Peninsula, a gap analysis of the Natura 2000 network for endemic Iberian and Balearic water beetles has been conducted to identify vulnerable areas (Sanchez-Fernandez *et al.*, 2008). Results show a good overlap of species distributions in mountainous areas, whereas the least overlap is found in river areas and stream environments, followed by lagoons, and ponds (Sanchez-Fernandez *et al.*, 2008). The study identifies a number of hotspots containing high-priority species, almost half of which are located outside the Natura 2000 network (Sanchez-Fernandez *et al.*, 2008). Furthermore, the existing network fails to capture the entire distributions of four species (*Iberoporus cermenius*, *Hydraena quetiae*, *L. monfortei* and *O. irenae*), with less than 40% of the distributions of nine further species being represented (Sanchez-Fernandez *et al.*, 2008).

One project found in this review which specifically assesses biodiversity adaptations for the future climate is the BRANCH (Biodiversity Requires Adaptation in Northwest Europe under a CHanging climate) Project (BRANCH, 2007). This project has planned a wildlife corridor in the area of Limburg, Netherlands, which will increase the connectivity between a variety of habitats including forest, heathland, marsh and arable land in the Dutch National Ecological Network and Natura 2000, in over 2,200 ha of planned habitat creation (BRANCH, 2007). The BRANCH Project models changes in habitat of the sand lizard, finding its distribution to become increasingly patchy as a result of climate change. Hence the creation of a wildlife corridor should reduce this impact (BRANCH, 2007). Another analysis carried out by the project, this time in the UK, identifies a need to increase the variety of habitats around the Hampshire heaths and grassland (BRANCH, 2007). Results suggest that only four of the existing lowland heath species are likely to maintain favourable climatic space in the future, and hence the ecological composition of this environment could undergo substantial changes by 2080 (BRANCH, 2007). It is therefore suggested that increasing the variety of habitats to support a changing assemblage of species is a favourable long-term option to help biodiversity here adapt to a changing climate (BRANCH, 2007).

In addition to the specific individual adaptation measures discussed above, a number of broader, collective measures are also being taken to aid species adaptation to climate change adaptation. DEFRA for example have developed a number of climate change adaptation principles as part of the England Biodiversity Strategy to act across the following sectors: agriculture; water and wetlands; woodland and forestry; towns, cities and development; coasts and seas (Smithers *et al.*, 2008). The five main principles of the strategy cover improving resilience; the integration of action across all sectors; the accommodation of change; and the development of knowledge, and strategic planning. The overriding principle, applying to all of the above is that action should be taken now, as a precautionary measure (Smithers *et al.*, 2008). Despite this emphasis, these appear from the literature search to be primarily theoretical principles, with few examples of action taken, although they can, and as seen below, are being used to inform adaptation practice (Smithers *et al.*, 2008).

Actors such as UKCIP are involved in numerous active projects to aid biodiversity adaptations to climate change (e.g. UKCIP 2010b, 2010c). In the UK Midlands for example, the West Midlands Biodiversity Partnership, the Environment Agency, and West Midlands Wildlife Trusts, in partnership with UKCIP, have developed a series of five adaptation principles (similar to those in Smithers *et al.*, 2008), as follows: (i) the development of ecologically resilient landscapes, (ii) the conservation of existing wildlife habitats and species, (iii) the reduction of human-induced stressors, (iv) sound decision-making based on analysis, and (v) the communication of the issues to policy-makers and the public (UKCIP, 2010b). In addition to this general policy, at a county level in Kent, UKCIP is working to develop a plan for future ecological networks in coordination with the BRANCH project, Natural England,

and Kent County Council, to form part of Kent County Council's Climate Change Programme (UKCIP, 2010c).

Natural England was found to be another key player as far as the UK is concerned, creating a number of local climate change frameworks and highlighting biodiversity adaptation in their report, 'Making space for wildlife in a changing climate' (Natural England, 2010). This report emphasises the importance of biodiversity, and its adaptation to climate change through reducing fragmentation, creating ecological networks and corridors, in addition to habitat restoration and creation. Natural England have assisted Tonbridge and Malling Borough Council to create a broad local climate change adaptation policy, including green infrastructure, increasing connectivity between ecosystems, and the creation of networks (Natural England, 2010). This framework will provide a multifunctional role, increasing aesthetics, and recreational opportunities in the area, whilst contributing to the achievement of both UK and Kent Biodiversity Action Plan targets (Natural England, 2010).

Finally, as far as future adaptation in Europe is concerned, integrated conservation strategies, like those above, with aspects of habitat connectivity management, assisted colonization, and restoration, are thought to be the best way to aid species adaptation to climate change, able to facilitate a wider range of species than one of the above alone (Loss *et al.*, 2011; Vitt *et al.*, 2010).

3.3 Coasts

The literature uncovered examples of a wide range of adaptation interventions; from traditional hard-engineering methods such as levees and embankments (van Dyke and Wasson, 2005), to breakwaters (Airoldi *et al.*, 2005; Guidetti *et al.*, 2005; Lamberti *et al.*, 2005), low crested structures (LCS) (Lamberti *et al.*, 2005; Martin *et al.*, 2005; Moschella *et al.*, 2005), seawalls (Blockley and Chapman, 2005; Bozek and Burdick, 2005; Glasby *et al.*, 2007) and soft-engineering approaches of beach nourishment (Lamberti *et al.*, 2005; Bishop *et al.*, 2006; Peterson *et al.*, 2006; Speybroeck *et al.*, 2006; Grippo *et al.*, 2007; Jackson *et al.*, 2010). The studies highlighted a number of high-impact schemes to protect low-lying land from tidal inundation, with a series of storm-surge barriers in the Netherlands (Noordwijk-Puijk *et al.*, 1979; Elgershuizen, 1981; Wolff, 1992; Schekkerman *et al.*, 1994; Smits *et al.*, 2006). The overall trend was a movement away from traditional hard engineering structures such as seawalls and embankments, towards a more dynamic coastal system which will provide the accommodation space needed for species to adapt to future sea-level rise. The majority of adaptation interventions can be covered under the terms managed realignment (Chang *et al.*, 2001; Townend and Pethick, 2002; Mazik *et al.*, 2007; Pontee, 2007; Andrews *et al.*, 2008; Reading *et al.*, 2008; Rotman *et al.*, 2008; Spencer *et al.*, 2008) and managed retreat (Maddrell, 1996; Emmerson *et al.*, 1997; French, 1999; MacLeod *et al.*, 1999; Marcus, 2000; Hazelden and Boorman, 2001; Lee, 2001). These were the two most popular adaptation interventions found in this review, and both include the removal or setback of previous hard defences, with methods such as de-poldering (de Ruig, 1998; Götting, 2001), de-embankment (Barkowski *et al.*, 2009; Kolditz *et al.*, 2009), and the breaching of dykes (Campbell and Bradfield, 1988; Bernhardt and Koch, 2003) being adopted. The net effect is one of coastal restoration, with the creation of saltmarsh and mudflats providing a sustainable, natural defence (Wells and Turpin, 1999; Warren *et al.*, 2002; Teal and Weishar, 2005; Darnell and Heilman, 2007; Verbessem *et al.*, 2007).

This review identified reference to two types of adaptation: planned, for example the construction of a sea wall, or the intentional breaching of defences in managed retreat; and autonomous. Autonomous adaptations included the unintentional breaching of defences due either to bad maintenance, or extreme weather events. An example of this unplanned adaptation was seen in the Netherlands, with a dyke breached after an extreme storm event in the Scheldt Estuary restoring tidal flow and transforming the area into brackish marsh (Eertman *et al.*, 2002). Another example from the Netherlands is presented by Bakker *et al.* (2002), where damages from winter storms were too costly to repair, and resulted in the breaching of summer dikes which were naturally restored to marshland.

Adaptation interventions are generally intended to impact on a particular sector, some examples being the construction of seawalls and breakwaters at Ria de Aviero, Portugal, to reduce beach erosion (da Silva and Duck, 2001); coastal wetland restoration schemes in the UK to offset habitat loss from coastal squeeze (Dixon *et al.*, 1998; MacLeod *et al.*, 1999; Pethick, 2002; WWF, 2002; Winn *et al.*, 2003;) and the construction of dams, sluices and storm-surge barriers in the Netherlands to reduce vulnerability to future sea-level rise and storm-surge events (Elgershuizen, 1981; Saeijs & Stortelder, 1982; Wolff, 1992; Schekkerman *et al.*, 1994; Smits *et al.*, 2006).

For the CLIMSAVE project, it is important to stress that the cross-sectoral nature of coastal adaptation measures means they will almost certainly impact on multiple sectors.

3.3.1 Wetland creation, managed retreat and managed realignment

Wetland creation, implemented for a variety of reasons, impacts mainly on coasts, by functioning as a natural defence (Tshirintzis *et al.*, 1996; Pethick, 2002; Darnell and Heilman, 2007; Rotman *et al.*, 2008) and on biodiversity by providing valuable habitat space (Mangin & Valdes, 2005; Desrochers *et al.*, 2008; Rotman *et al.*, 2008; van Proosdij *et al.*, 2010). This was the most common cross-sectoral linkage identified in the literature, with wetland restoration often part of managed realignment and retreat schemes.

Coastal wetlands provide the services of wave dampening, and the protection of existing seawall defences by the prevention of scour (Hazelden & Boorman, 2001; Hofstede, 2003; Andrews *et al.*, 2008). One study reported an increasing decline in average wave-height after tidal restoration, but found that this reaches an upper limit after three seasons (Roman *et al.*, 2002). Möller *et al.* (2001) consider the effectiveness of saltmarsh as a coastal defence to be particularly high in shallow water conditions, calculating that the removal of marsh in water less than 0.7 m deep would result in a three-fold increase in average wave-height. The establishment of saltmarsh vegetation also reduces erosion rates by creating an effective sediment sink (Hinkle and Mitsch, 2005; Mazik *et al.*, 2007; Rotman *et al.*, 2008). To quantify this effect, one study calculated that the intertidal area created by managed realignment on the Humber would result in the annual accretion of 1.2×10^5 tonnes of sediment and reduce local erosion rates (Andrews *et al.*, 2008). The realignment of hard defences is also known to enhance the storage capacity of an estuary, hence reducing vulnerability to storm-surge events (Klein and Bateman, 1998; Eertman *et al.*, 2002; Jickells *et al.*, 2003; Pontee *et al.*, 2006; Dixon *et al.*, 2008).

On the other hand, restoration schemes and the realignment of current defences can also impact negatively on the coastal sector, with an enlarged tidal prism leading to high rates of erosion as the site re-equilibrates with its surroundings (Emmerson *et al.*, 1997; Marcus, 2000; Bakker *et al.*, 2002; Hofstede, 2003; Symonds and Collins, 2007b; Verbessem *et al.*, 2007). Two specific examples of this in the literature were provided by US studies, and are used to highlight this negative effect. The first was seen after habitat restoration in California, where the restoration of tidal influence to 120 ha of land resulted in a 30% increase in tidal prism, with increased tidal velocities causing creeks and channels to erode, and inducing a positive feedback effect (van Dyke and Wasson, 2005). The second was reported by van Proosdij *et al.* (2010) with localised erosion after saltmarsh restoration in the Bay of Fundy causing the head of a tidal channel to retreat by 35 m.

3.3.2 Storm-surge barriers and the Delta Project

Large-scale engineering projects, such as the storm-surge barriers in the Netherlands were shown to impact over a range of sectors.

The Delta Project led to the closure of numerous estuaries along the Dutch coast, and has had the desired effect with respect to reinforcement of the coastline and protection against flooding (Wolff, 1992). In contrast, the closure of estuaries has led to a subsequent reduction of be 120 km^2 in tidal area on Eastern Scheldt, causing many intertidal zones to dry out (Elgershuizen, 1981; Saeijs and Stortelder, 1982; Smits *et al.*, 2006). The new hydrodynamic regime has resulted in widespread erosion, with a reported loss of 120 million m^3 of sediment from the Oosterschelde tidal basin, and a doubling in the rate of cliff retreat since the completion of the project (Louters *et al.*, 1998). Research shows that the Delta Project will cause the future loss of all tidal flats in the area (Smits *et al.*, 2006), and therefore could be seen as an example of maladaptation, with hard-engineering damaging natural coastal defences and increasing rates of erosion as a result of a progressively more unnatural regime.

3.3.3 Low Crested Structures (LCS)

LCS are generally seen as an effective coastal defence, with structures such as breakwaters and seawalls reducing the amount of wave energy reaching the shore (Marcus, 2000; Airoldi *et al.*, 2005; van Dyke and Wasson, 2005; Lamberti *et al.*, 2005) and local erosion rates, as well as accreting sediment on their leeward side (Airoldi *et al.*, 2005; Bozek and Burdick, 2005; Lamberti *et al.*, 2005; Martin *et al.*, 2005). An example of this is the installation of a breakwater system at Lønstrup, Denmark, which decreased the amount of wave energy reaching the shoreline, and as a result prevented further erosion of the cliffs behind the beach (Lamberti *et al.*, 2005).

In contrast to these local benefits, many reports identify increases in erosion downstream, after the sediment supply to these sites has been reduced. Lamberti *et al.* (2005) report submerged groynes and a breakwater in Italy to have reduced the supply of sediment to an area downdrift, causing an entire beach to erode. As a result, enhanced local defences have led to a decreased amount of protection elsewhere. LCS have also been reported to alter local bathymetry and hydrodynamics (da Silva and Duck, 2001; Lamberti *et al.*, 2005; Martin *et al.*, 2005). One example of these changes is the observed increase in current strength and subsequent increases in water depth (by as much as 10 m) after the installation of breakwaters in Portugal (da Silva and Duck, 2001).

Embankments have been reported to have similar adverse impacts on the coastal system, with increases in coastal hydrodynamics, storm-surge height and downstream erosion (Reise, 1998; 2005). Von Storch *et al.* (2008) have found that in contrast to reducing vulnerability to storm-surges; the frequency and intensity of surges, along with mean tidal high water in the Wadden Sea area has actually increased since the 1970s. The study concludes that 75% of this increase is due to the installation of hard-coastal defences along estuaries, which decrease bed roughness and the efficiency of which the estuary dissipates incoming wave energy.

3.4 Forests

Climate change could damage forest ecosystems in a number of ways including direct impacts on changing environmental conditions and indirectly through fire, infestation, disease and windthrow (Sedjo, 1991). Forests could adapt naturally as they have done in the past leading to changes in the ranges of important tree species, but a critical issue is the rate at which tree species would migrate under global warming (Sedjo, 2010). It is customary to classify adaptation measures into: anticipatory, reactive, autonomous and planned. Anticipatory, also referred to as proactive adaptation takes place before impacts of climate change are observed. Early warning systems to prepare for forest fires are a classical example of anticipatory adaptation. Reactive adaptation is that which takes place after the impacts of climate change have been observed (Robledo *et al.*, 2005). For example, salvage logging after a storm (Garforth, 2012) or after a fire (Sedjo, 2010). Behavioural changes taken by private actors as a reaction to actual or expected climate change are known as "autonomous" adaptation. For example, the change of date of planting /seeding and harvesting by the farmer, due to a change in rainfall patterns (Holmgren *et al.*, 2007). Planned adaptation is the result of a deliberate policy decision based on an awareness that conditions have changed and that action is required to return to, maintain, or achieve a desired state (Robledo *et al.*, 2005). For example, tree planting and the monitoring of forests (Bernier and Schoene, 2009). Literature on the forestry sector has a high focus on regions such as the tropics, which fall outside the study area. However, where these are valuable examples, they have been included in this review.

Adaptation measures emphasized in the literature are: afforestation, reforestation and agroforestry.

3.4.1 Afforestation

According to Sedjo (2010), in 2009 there were 45,083 ha planted forests in China; 32,578 ha in Japan; 17,340 ha in Russia; 16,238 ha in United States; 10,682 in India; 9,871 ha in Indonesia; 4,892 ha in Brazil; 4,425 ha in Ukraine; and 2,284 ha in Iran. In southern Africa, the afforestation rate is around 11,000 hectares per year. Examples of adaptation measures include: the afforestation of areas to protect against drought and aridity and provide firewood, fodder, tannin, pulpwood, shelterbelts and soil improvement (UNFCCC 2008a), the planting of trees in Tajikistan to protect from erratic rainfall and stabilise eroding soils and slopes (UNFCCC 2008b) and the Five-year Action Plan for Mangrove Management in the Gulf of Thailand, which preserves mangrove forests and promotes sustainable use of mangrove resources (UNFCCC 2008c).

3.4.2 Reforestation

In Brazil, Rio de Janeiro, Parana and Santa Catarina states were once completely covered with Atlantic rainforests. Most of this indigenous ecosystem has been destroyed. About 30,000 seedlings of *Araucaria angustifolia* and *Ilex paraguayensis (Yerba matı)* were planted by graduate students from Rottenburg. The 5,000 ha area was named Pro-Mata. The concept, then new, was to plant young trees right into existing secondary vegetation which consisted mainly of *baccharis* bushes and mimosa trees. During the last 12 years the young *Araucaria* trees reached heights of five metres and more. This will diminish the impact of frost and drought on agriculture, especially reducing harm to coffee and citrus crops (Bodegom *et al.*, 2009). Other examples are the reforestation of Mount Malindang, in Philippines which started in 2008 (Bodegom *et al.*, 2009), the reforestation of mangroves forests in the Philippines that began in 1930-1950 (Primavera and Esteban, 2008), the reforestation of mangroves in Malaysia, Florida, Panama, Kenya, Hawaii, Fiji and Burma, and afforestation and reforestation in areas of mangroves in Bangladesh, India, and Vietnam (Kairo *et al.*, 2001).

3.4.3 Agroforestry

The following are global examples of adaptation in agroforestry:

1) The growing of appropriate tree species on cultivated land to reduce vulnerability to hurricanes and to provide various other benefits, including a reversal of the deforestation trend in Grenada (World Agroforestry Centre, 2007).

2) Deep-rooted trees are used in agro-forestry operations in order to tap more moisture from a lower depth during the dry season, so as to increase the overall productivity of land in Zimbabwe (and elsewhere). Different crop canopies use light efficiently, and the agro-forestry systems return large amounts of nutrients to the soil, as well as provide shelter against wind erosion (Agobia, 1999).

3) The cultivation of drought-tolerant fruit trees to diversify household income sources, ensure food security and provide shade and fuelwood in Bangladesh (Selvaraju *et al.*, 2006).

4) The alley cropping (the practice of planting trees in rows with food or cash crops between them) which is used to reduce the vulnerability of the population and their environment to hurricanes and hurricane-related devastations, in Jamaica (Thomas-Hope and Spence, 2002).

5) The cultivation of moringa trees that are very drought-resistant and tolerate a wide variety of soil types in Senegal. They can be used to combat malnutrition by providing enriched food and by treating drinking water (Boven and Morohashi, 2002).

6) Finally, in the Himalayas, in India, where communities are faced with erratic rainfall during spring and summer, farmers have developed agro-forestry practices to ensure food security and additional income, particularly growing cardamom, bamboo groves and fruit trees (Verma 1998; Seppala *et al.*, 2009).

3.4.4 Tending and thinning

In Europe, the key adaptation measures are afforestation and reforestation, through enhancing natural regeneration, planting seedlings, or seeding, thinning and harvesting practices (Kolström *et al.*, 2011). The pan-European assessment ‘SilviStrat’ (Kellomäki and Leinonen, 2005) explored the impacts of climate change on forest productivity, carbon storage and biodiversity. In general they found that forest productivity and carbon storage increased in northern and central Europe but declined in southern Europe due to drought impacts. They recommended planting new species that are drought-tolerant and frost-tolerant as an adaptation strategy. They also recommended increased intensity of thinning in areas where productivity was likely to increase (Johnston *et al.*, 2010), as the more diverse and larger the seedling population is, the greater the potential for populations to adapt to environmental changes. Tending and thinning can also help to manage increasingly mal-adapted stands in a changing environment (Kolström *et al.*, 2011; Lindner *et al.*, 2008).

3.5 Urban

Over two thirds of Europeans live in urban areas (European Commission, 2011), although they only cover 1.5% of Europe (PELCOM, 2000), they can play an important role in both adaptation and mitigation. Action 6 of the Biodiversity Strategy states that the European Commission will "*develop a Green Infrastructure Strategy by 2012 to promote the deployment of green infrastructure in the EU in urban and rural areas"* and green infrastructure, as is shown below, can provide an important means of adaptation in urban areas. The same is true for mitigation, as, although CO_2 emissions per person are much lower in urban areas, their density can make for more energy-efficient forms of housing, transport and service provision, meaning that mitigation actions may be more efficient and cost-effective (European Commission, 2011).

3.5.1 Urban greenspace

The provision of urban greenspace, which contributes to green infrastructure, is associated with a number of ecosystem services, including reductions in surface runoff, climate regulation, and carbon storage. Greenspace therefore has the potential to be used both as a form of climate change adaptation and mitigation. This will be briefly discussed, before the effectiveness of two specific measures to increase urban greenspace; green roofs and urban trees, is reviewed in detail.

As previously mentioned, urban greenspace is able to influence the local climate; reducing local surface temperatures by shading, and reducing air temperatures through evaporative cooling and albedo effects (e.g. Gill *et al.*, 2007). Increasing the provision of greenspace therefore has the potential to ameliorate the temperature of urban areas under climate change (e.g. Bowler *et al.*, 2010). A meta-analysis of the cooling potential of urban parks found they were able to reduce ambient daytime temperature by an average of 0.94°C; with an average night-time reduction of 1.15°C (Bowler *et al.*, 2010). A quantitative modelling study found that increasing the green area by 10% in dense urban areas of Greater Manchester could retain maximum surface temperatures at, or below the 1961-1990 baseline until the 2080s for all emissions scenarios, mitigating the effects of climate change (Gill *et al.*, 2007). Urban greening can also provide regional cooling benefits in the city of Manchester, with grass being the most effective at reducing peak temperatures, achieving reductions of up to 24°C, compared to the maximum 19°C reduction provided by tree shade in the city (Armson *et al.*, 2012). Grass surfaces are also associated with substantially lower surface temperatures than concrete, which has surface temperatures 17°C and 4°C above peak air temperature in direct sunlight and shade respectively (Armson *et al.*, 2012). This, compared to grass which saw maximum temperatures 1°C and 4°C below peak air temperatures in sun and shade respectively, highlights the ability of greening measures to considerably improve the urban climate and mitigate the urban heat island (Armson *et al.*, 2012). As a result of this potential, substantial greening is taking place in the city of Transvaal, Denmark, where developers aim for all dwellings to be located within 200 m of greenspace (Kleerekoper et al., 2012).

In an attempt to increase the provisioning of greenspace in Manchester, a programme called the Corridor Partnership has been created to green the Oxford Road corridor, an area of 2.73 km^2 close to the city centre (Kazmierczak *et al.*, 2010). This is a major transport corridor in the city which currently has a green space provision of only 15% (Kazmierczak *et al.*, 2010). Increasing the amount of green space in this area has the potential to decrease surface temperatures under high emissions scenarios by 3.0-4.9°C, and by 3.8-5.2°C under low emissions scenarios (both ranges quoted as degrees below the 1961-1990 baseline)

(Kazmierczak *et al.*, 2010). This would significantly reduce the need for artificial cooling in buildings during the summer months and hence contribute indirectly to mitigation through avoided or reduced emissions (Kazmierczak *et al.*, 2010).

Increasing the amount of urban greenspace has additional benefits, including improved human comfort by reducing the effects of heat-stress; air quality improvements (Clark *et al.*, 2008; Fioretti *et al.*, 2010; Ottelé *et al.*, 2011); aesthetics (Rotherham, 2010; Berkooz, 2011); and a reduction in surface runoff volumes (McPherson & Rowntree, 1993; Asadian & Weiler, 2009; Bowler *et al.*, 2010). In the Greater Manchester area however, modelling results show that the predicted future increases in winter precipitation will increase runoff regardless of greening (Gill *et al.*, 2007). As a result, increased storage for stormwater will be required in addition to the provisioning of new greenspace (Gill *et al.*, 2007).

It is important to note that urban greenspace does not increase ecosystem services in all cases, for example, plantings which increase allergens, promote, or host invasive species can provide a disservice (Pataki et al., 2011). Furthermore, projected increases in the number of consecutive dry days and heat waves during summer months may counteract this the cooling effect provided by greenspace, as for example, when grass becomes dry it loses the ability for evaporative cooling (Gill *et al.*, 2007; Armson *et al.*, 2012). As future water pressures may result in conflict for use of water for irrigation, it is important to develop sustainable irrigation measures for greenspace, for example by rainwater harvesting, the re-use of greywater, and floodwater storage to ensure that they continue to regulate urban climate (Gill *et al.*, 2007).

Green roofs and urban trees have both adaptation and mitigation potential, but the latter is discussed under urban mitigation in Sections 4.5.2 and 4.5.1 respectively.

3.5.2 Green roofs

Air temperature reductions

It is the evaporative cooling effect associated with the vegetation on green roofs which reduces ambient air temperature (Lundholm *et al.*, 2010), an effect measured in Japan to provide around 5°C cooling (Onmura et al., 2001). This ability of green roofs enables them to ameliorate the urban climate, improve the internal comfort of buildings and also to mitigate the effects of the urban heat island (e.g. Alexandri and Jones, 2008; Fioretti *et al.*, 2010; Scherba *et al.*, 2011).

A quantitative modelling study concerning the potential of green roofs and walls to reduce the urban heat island effect in a selection of European cities found this to be significant (Alexandri and Jones, 2008). The magnitude of temperature reduction varied with location, being the most effective solutions for mitigating the urban heat island in central and southern areas of the Mediterranean basin (Zinzi and Agnoli, 2011), with the substantial daytime temperature reductions experienced in Athens comparable to the greatest maximum temperature decrease of 26.1°C simulated for Mumbai (Alexandri and Jones, 2008). In contrast, green roofs have less potential to decrease temperatures in areas with cooler climates such as London and Moscow, which saw the smallest decreases in average day-time temperature of only 9.11°C (Alexandri and Jones, 2008). In addition to lowering roof surface temperatures, the heat accumulated by green roof systems through the course of a day is slowly released during the night, found to result in large reductions in peak temperature fluctuations (Scherba *et al.*, 2011).

The capacity of green roof systems to ameliorate urban temperature is dependent on a number of environmental factors. In order to explore these using the current literature, it was necessary to widen the study area, using examples from the US and Canada. Firstly, results from Halifax, Canada, reveal that, if medium-only roof modules are grown, temperatures can be reduced by over 10°C, which is considerably higher than the average 2°C reduction achieved with monocultures and one life-form groups in the study (Lundholm *et al.*, 2010). Roofs planted with species of higher albedo and richness as well as a low biomass variability have been found to achieve the largest temperature reductions, and therefore if the highest temperature mitigation is to be achieved, species such as *S.bicolor* should be planted (Lundholm *et al.*, 2010). As far as the rate of cooling is concerned, results from green roof in Oregon show that mosses are associated with faster rates of cooling of the roof surface than can be achieved by a medium-only module, with rates of 6°C h^{-1} and 1.1°C h^{-1} respectively (Anderson *et al.*, 2010). Moisture also has a large role in determining the performance of green roofs, with a well wet roof having a high cooling performance (Zinzi and Agnoli, 2011). Therefore, under climates with hot and dry summers such as the Mediterranean, water for roofs may need to be managed; calibrated to the prevailing climatic conditions and energy usage (Zinzi and Agnoli, 2011).

Roof surface cooling

The passive cooling associated with green roofs has a substantial affect on roof surface temperature, able to reduce the amplitude of roof-slab temperature fluctuations in France by as much as 30°C (Jaffal *et al.*, 2012). This reduction in temperature results from the process of evapotranspiration, and the higher albedo of green roofs compared to conventional roofing systems. The increase in albedo associated with the installation of green roofs in a neighbourhood in the Ukraine has been measured by LANDSAT satellite data at around 0.07 (Mackey *et al.*, 2012). The result of these roof systems having a higher albedo is that they absorb less solar radiation, reducing surface temperature and heat-flux into the building (Lundholm *et al.*, 2010). It is important to note that the extent to which green roofs increase albedo is dependent on the species used and hence their albedo can be highly variable (Lundholm *et al.*, 2010). For example a study of green roofs in Chicago found those with medium-only modules had an albedo of 0.158, while those with vegetated modules a higher albedo of 0.180-0.195 (Lundholm *et al.*, 2010). Both these values are considerably higher than the 0.0666 ± 0.006 associated with conventional roofing systems (Lundholm *et al.*, 2010) and albedo values should not particularly differ from those in Europe.

Increased energy efficiency

The soil and vegetation layers of green roofing systems provide extra insulation for the building; preventing solar radiation from reaching the building skin during the summer, and conversely the escape of internal heat during winter (Ottelé *et al.*, 2011; Zinzi and Agnoli, 2011). Furthermore, green roof systems have a high energy efficiency; the vegetative layer acting as a buffer against the wind, which for a conventional roofing system can reduce energy efficiency by as much as 50% (Ottelé *et al.*, 2011; Perini *et al.*, 2011). Energy savings arising from green roofs are estimated at around 10-15% (Bigham, 2011), with a 12% reduction in energy demand reported for a green roof in the Mediterranean region (Zinzi and Agnoli, 2011), and a 6% reduction in annual average energy demand for a single family house in France (Jaffal *et al.*, 2012). Green roofs can be very effective in cities such as Athens, able to reduce high cooling loads in buildings by 66%, with 4-hour reductions in cooling energy demand being reported (Alexandri and Jones, 2008). Reductions here, and in other parts of the Mediterranean are highest during the summer, and could potentially remove the need for cooling systems in these regions (Zinzi and Agnoli, 2011).

In contrast, in colder climates, such Moscow, the use of evergreen species on the roof can provide an extra layer of insulation, contributing to wintertime energy savings as a result of reduced heat loss from buildings (Perini *et al.*, 2011). However, it should be noted that on non-insulated simple buildings, these species become frozen in winter, no-longer providing an insulative benefit (Teemusk and Mander, 2010). The opposite was found to be true during winter in the Mediteranneam, with modelling results showing a 16% increase in heating load on the top floor of a building in Athens during December (Spala *et al.*, 2008). The costs benefits associated with these energy savings is discussed in Section 15.1.5

Stormwater management

The vegetation in green roof systems is able to retain between 25 and 100% of rainfall, reducing peak and total runoff, as well as returning a considerable amount of the precipitated water to the atmosphere via evapotranspiration (Oberndorfer *et al.*, 2007). Many studies found in this review acknowledge these benefits of green roof systems, but fail to consider them as a form of climate change adaptation (e.g. Stovin *et al.*, 2012).

To demonstrate the effectiveness of green roofs in reducing runoff, reductions in peak flow of 74 ± 20% were measured for roofs in central and north-western Italy during the autumn and winter seasons (Fioretti *et al.*, 2010). In addition, green roofs were measured to delay peak flow by over 2 hours, with the vegetation able to retain an average of 23 ± 31% of precipitation (Fioretti *et al.*, 2010). This potential for stormwater management appears to be comparatively lower in Manchester, where green roof systems were found to reduce runoff from an 18 mm rainfall event by only 17-19.9% (Gill *et al.*, 2007). In Brussels, a modelling study examining the effectiveness of green roof systems as a form of climate change adaptation found that the extensive use of these systems on 10% of the current building stock would reduce runoff in the region by 2.7%, and by 54% on an individual building basis (Mentens *et al.*, 2006)

Factors influencing the stormwater alleviation potential of green roofs include the amount that plants transpire, those with a greater canopy biomass providing a larger total area for gas exchange (Lundholm *et al.*, 2010). In addition, roofs covered in mosses such as *Racomitrium canescens* were found to have a 12-24% higher stormwater retention than vascular or medium-only roofs, for example being able to hold 47 L m^{-2} without any medium, compared to water storage of 33 L m^{-2} by a roof with a 2.5 cm deep layer of medium (Anderson *et al.*, 2010).

Implementation in Europe

Green roofs have been used extensively in Augustenborg, a neighbourhood in the city of Malmö, where they have been constructed on all new-builds since 1998 (Kazmierczak and Carter, 2010). The neighbourhood now has a total of over 30 green roofs constructed by the MKB social housing association, covering an area of 2,100 m^2 (Kazmierczak and Carter, 2010). In addition, a Botanical Roof Garden has also been built on an old industrial roof in the city covering an area of 9,500 m^2, making it the largest green roof in Scandinavia (Kazmierczak and Carter, 2010). The project was funded by the Swedish Department of the Environment, and the EU programme, LIFE at a total of SEK 10 million (Kazmierczak and Carter, 2010).

In the city of Basel, Switzerland, regulation makes the installation of green roofs mandatory on all new buildings with flat roofs as part of the city's strategy for biodiversity (Brenneisen, 2006). In addition, the city of Transvaal, Denmark, also has large potential for the use of green roofs, as over 95% of its buildings have a flat roof (Kleerekoper *et al.*, 2012). The construction of green roofs in the city would have many benefits, including the provision of additional space, and the use of roofs as garden areas, as well as improving the appearance of the city (Kleerekoper *et al.*, 2012).

3.5.3 Urban trees

<u>Temperature reductions</u>

Urban trees function in two ways to reduce temperature. Locally, they provide shade reducing the amount of short wave solar radiation reaching the surface. Secondly, on a more global scale they provide direct cooling as a result of evapotranspiration (Armson *et al.*, 2012). They also sequester carbon, thus contributing to climate change mitigation and this is discussed in Section 7.2.3.

Numerous studies in the US have examined urban tree planting as a way to reduce the risk of warming, identifying the potential for large local temperature decreases (e.g. McPherson and Simpson, 2003; Armson *et al.*, 2012). In Europe, a program called INTEgrative Research on Forest Areas, Citizens and urban Environment has investigated the use of green space and street trees for heat stress mitigation (Lafortezza *et al.*, 2009). The program examined the cities of Gateshead (UK), Milan (northern Italy) and Bari (northern Italy), finding that trees in the urban area were able to ameliorate the urban microclimate, provide shading and reduce the frequency of heat stress events (Lafortezza *et al.*, 2009). A similar capacity has also been found in Manchester, where tree planting in residential gardens has the potential to reduce peak surface temperatures by between 0.5 and 2.3°C (Hall *et al.*, 2012). However, the ability to increase tree cover here appears fairly limited, in the range of 2.8-5.3% only, making it impossible for urban trees to maintain temperatures at current levels beyond the 2020s, with at most a reduction of 2.3°C (Hall *et al.*, 2012). In London to reduce surface temperatures and ameliorate the future climate, the Greater London Authority aim to increase tree-cover across the city from 20% in 2009, to 25% in 2025; an increase which will require the planting of an extra 2 million trees (GLA, 2010).

As far as management is concerned, there is a need to plant species resilient to drought to ensure that they continue to regulate local conditions under a warming climate (Gill *et al.*, 2007). Small-leaved tree species have been identified as suitable for locations where high air temperatures are predicted to occur more frequently as a result of climate change; with species such as *Gleditsia triacanthos* able to maintain a relatively constant temperature in foliage even at high ambient temperatures, therefore having an advantage when not irrigated (Leuzinger *et al.*, 2010).

3.5.4 White-topping and cool paving

White-topping increases the albedo of urban areas, and can be applied to surfaces such as roofs and pavements (often termed 'cool paving'). These methods mitigate climate change in the long-term by reducing temperatures over decades to centuries (Akbari *et al.*, 2012). A global simulation study of the area between 20° and 45° latitude has explored the potential of white rooftops and light-coloured pavements to increase the albedo in cities (Akbari *et al.*, 2012). The simulations suggest that 'whiting' could increase the albedo of urban areas by around 0.1, and in terms of climate change mitigation, if the albedo of 1 m^2 surface is increased by 0.01, the resultant long-term cooling effect is estimated at 3 x 10^{-15} K, consistent with a 7 kg reduction in CO_2 emissions (Akbari *et al.*, 2012). It is estimated that if whiting is employed in all urban areas, with the range of global cooling estimates in the study, this could amount to temperature reduction equivalent to 25-150 billion tonnes CO_2 (Akbari *et al.*, 2012).

An example of white-topping in Europe is the installation of white reflective roofs in the city of Ondiep, Denmark as a form of "climate proofing" (Kleerekoper *et al.*, 2012). These surfaces reflect sunlight and are able to maintain a high albedo, being designed to repel dirt (Kleerekoper *et al.*, 2012). Reflective roof surfaces have also been used in the city of Chicago, US, and have increased the city's albedo by 0.016, being the preferred option to reduce the urban heat island effect (Mackey *et al.*, 2012). Reflective roofs here have are associated with a stronger LANDSAT cooling than green roofs (Mackey *et al.*, 2012).

White-topping has also been considered for the city of Athens, with test studies used to determine the albedo of various coloured surfaces (Synnefa *et al.*, 2011). Results show that off-white asphalt has the best potential to cool the surface temperature, having the highest solar reflectance of all materials tested at 0.55, and being over 6°C cooler than the black conventional asphalt surface (Synnefa *et al.*, 2011). As well as reducing average surface temperature, off-white asphalt substantially reduced the mean maximum diurnal surface temperature range (this being 48°C instead of the 60°C of the black conventional asphalt) (Synnefa *et al.*, 2011). Cool paving using concrete light yellow blocks, which have a high solar reflectance of around 60%, has been installed in one of the city's urban parks with the aim of improving the urban microclimate (Santamouris *et al.*, 2012). This cool pavement has a high cooling performance, reducing surface temperature in the park by 12 K, although the cooling effect is substantially reduced under cloud cover, comparable to that of traditional materials (Santamouris *et al.*, 2012). In total, an area of 4,500 m^2 in Athens has been paved with reflective materials, making it the most large scale application of cool pavements in any urban area, reducing temperatures and mitigating climate change (Santamouris *et al.*, 2012).

3.5.5 Rainwater harvesting and greywater re-use

Rainwater harvesting and greywater re-use are two methods which can be used to reduce the impacts of drought under climate change. Rainwater harvesting systems (RWHS) collect water from the runoff of impervious surfaces such as roofs and urban catchments (Graddon *et al.*, 2011), whereas greywater is that collected and recycled from washing operations, for example kitchen sinks, showers and bathtubs (Memon *et al.*, 2005; Li *et al.*, 2010; Domènech and Sauri, 2011). Once the water is collected, it can be used for a variety of purposes, with rainwater commonly used for high quality applications such as landscape irrigation. Greywater in contrast is of a much lower quality, containing dissolved contaminants from detergent and soap products, and is therefore used for low-quality water applications only, including toilet flushing, laundry, and car washing, although harvested rainwater can also be used for these purposes (Li *et al.*, 2010; Domènech and Sauri, 2011). These methods are most effective at the neighbourhood rather than regional scale (Farreny *et al.*, 2011), both decentralising the water supply, reducing potable water use (Wise *et al.*, 2010) and increasing regional resilience to drought by improving water security (Graddon *et al.*, 2011).

These systems have large potential. For example in Ireland, it is estimated that a combination of RWHS and greywater re-use have the capacity to meet almost 94% of household domestic water demands (Li *et al.*, 2010). Densely populated areas of the Mediterranean with a high number of roofs concentrated in a small area could also benefit (Farreny *et al.*, 2011). For example, in the Spanish municipality of Sant Cugat del Vallès, a rooftop RWHS has the potential to supply 16% of the towns total domestic water demand, and by diversifying water supply, the municipality is able to become more self-sufficient (Domènech *et al.*, 2011; Domènech and Sauri, 2011). The municipality was the first in Spain to change the building code, mandating all buildings with over 300 m^2 garden to install a RWHS through local regulations (Domènech *et al.*, 2011). In addition, since 2002 all newly built dwellings with over eight apartments, or an annual shower water consumption of over 400 m^3, are required to install a rooftop RWHS to re-use the greywater from the shower for toilet flushing (Domènech and Sauri, 2011). Research has shown that a water tank of 70 m^3 volume would be sufficient to irrigate a communal garden of 300 m^2 (Domènech and Sauri, 2011). In addition, a tank of 6 m^3 in a single family house would be able to supply 100% of the laundry water requirements, with water savings of 16 litres per capita per day (Domènech and Sauri, 2011). After the success of Sant Cugat del Vallès RWHS, the uptake of water recycling systems in Spain has increased, with over 40 municipalities in the region of Catalonia enforcing local regulations to encourage the installation of these systems in new buildings (Domènech and Sauri, 2011). The economics of RWHS is discussed in Section 15.1.5.

Decentralising the water supply has other benefits; both giving control to individuals, increasing awareness of their water consumption (Domènech *et al.*, 2011), and allowing homes to become self-sufficient if the public water supply is interrupted (Li *et al.*, 2010). As well as increasing water security, RWHS have the potential to lower flood risk by reducing the volume of runoff in urban areas, and are hence sometimes termed 'preventative systems' (Li *et al.*, 2010). These systems can, however, be costly with a long pay back period (see Section 15.1.5).

3.5.6 Flood protection

With respect to water resources, the built environment is more likely to adapt to climate change, rather than offer climate mitigation measures. These areas are often protected by hard structures against rising water levels, but other methods of adapting, such as allowing greater water infiltration, water flow and storage (including reduced water abstraction) are becoming more common. Flood defences can also be seen as innovative if they are integrated into the existing landscape. For instance, where necessary, hard defences in urban areas and emergency drainage channels can reduce flood risk. Near Maidenhead, UK, the Jubilee Flood Alleviation Channel was constructed in the late 1990s/early 2000s. The 11 km long channel acts as a relief channel to the River Thames. When river levels are predicted to be high, water is diverted from the Thames, thereby increasing the river storage area. The scheme cost £110 million to build and protects against the 1-in-67 year flood (Hansford, 2004). Protecting the built environment does not just mean adapting in the urban area. Howgate and Kenyon (2009) describe a study where Scottish farmers upstream of a town set aside their tenure to be flooded, rather than the town further downstream. This brings dual benefits of less hard defences and engineering, which comes at a lower cost to the tax payer.

3.5.7 Sustainable Urban Drainage Systems

Many of the greening measures discussed earlier in Sections 3.5.1-3.5.3 of this report can be implemented as a part of a Sustainable Urban Drainage System (SUDS). Additional measures implemented under SUDS include the construction of swales, stormwater detention ponds, and permeable pavements. These aim to reduce the effects of runoff by increasing stormwater retention and delaying peak runoff to reduce both the current and future risk of urban flooding (Wise *et al.*, 2010). Little was found in the literature review concerning the use of permeable paving in Europe as a climate change adaptation strategy. Permeable pavements have however been modelled in a car park in the UK Midlands, with results showing that this measure has the potential to store around 55% from a 15 mm h^{-1} storm event (Andersen *et al.*, 1999). The potential benefits of pervious pavements are also being examined in Santander, northern Spain, as part of SUDS (Gomez-Ullate *et al.*, 2011). These surfaces provide multiple services; both filtering and storing runoff to reduce urban flooding events, whilst increasing water quality (Gomez-Ullate *et al.*, 2011). Over time, the amount of water stored in pervious pavement systems was modelled to be sufficient to irrigate a 10 m^2 garden for almost a month-long period of drought (Gomez-Ullate *et al.*, 2011).

SUDS are needed for a variety of reasons. Firstly, existing drainage systems in many old urban areas with combined sewer systems, such as the city of Odense, Denmark, are already vulnerable to flooding and prone to overflowing during heavy rainfall events (Semadeni-Davies *et al.*, 2008a; Fryd *et al.*, 2010). SUDS disconnects stormwater flows from combined sewer systems to minimize the number of overflow events, and with projected increases in precipitation and urbanisation, sustainable drainage systems are needed to reduce stormwater flows and runoff rates (Semadeni-Davies *et al.*, 2008a). Stalenberg (2012) and results from the EU GRaBs[6] project illustrate examples of multifunctional SUDS measures to reduce urban flooding with e.g. underground parking facilities, set back banks, buildings raised and overhanging the river, green roofs, community gardens and water banking.

Research based on climate and urbanisation scenarios highlights the need of SUDS in areas such as Sweden where future urbanisation could increase stormwater volumes by 75%, which in combination with the effects from climate change is predicted to increase stormwater volume flows by up to 450% (Semadeni-Davies *et al.*, 2008a). Even neglecting the impact of future urbanisation, increases in precipitation are likely to worsen current drainage problems in towns such as Helsningborg, Sweden (Semadeni-Davies *et al.*, 2008b). As a result, SUDS are seen as the best option to reduce runoff response in urban areas such as the city of Malmö, and although in many cases such systems are not installed for the prime purpose of climate change adaptation, they do reduce urban flooding; a phenomenon expected to increase under climate change (Villarreal *et al.*, 2004; Kazmierczak and Carter, 2010). In Malmö impervious areas have been disconnected from the combined sewer, connected instead to a new open stormwater system which channels stormwater runoff through a series of swales, green roofs, ponds, channels and small wetlands (Villarreal *et al.*, 2004). This scheme was designed to reduce urban flooding by 70%, and the ponds are designed to delay peak storm flows for a 10-year rainfall event; storing water and helping to regulate local discharge (Villarreal *et al.*, 2004; Kazmierczak and Carter, 2010). Part of the scheme includes a SUDS-based retrofitting project in the Augustenborg neighbourhood, funded by a SEK 2.2 million government grant (Villarreal *et al.*, 2004). Green roofs in this neighbourhood intercept around half of the total annual runoff (Kazmierczak and Carter, 2010), and are planted with drought tolerant varieties such as sedum moss to retain functioning under future climatic conditions (Villarreal *et al.*, 2004).

Sustainable drainage systems are also seen as an appropriate adaptation to climate change in the Valencia region of Spain, where two towns, Xativa and Benaguasil, are at risk from seasonal flooding from heavy rainfall events (Casal-Campos *et al.*, 2012). This development

[6] Green and Blue Space. Adaptation for Urban Areas and Ecotowns. http://www.grabs-eu.org/

is part of the EU LIFE+ Project, AQUAVAL, and will consist of filter trenches, infiltration basins, dry swales, subsurface storage, green roofs and RWHS, significantly increasing the drainage capacity of the area under future precipitation scenarios (Casal-Campos *et al.*, 2012).

SUDS appear to be widely implemented in the city of Glasgow, in areas such as the Belvidere Hospital, Celtic FC Stadium and the Pollok Centre as part of the City Council's Glasgow Surface Water Management Project (Scholz *et al.*, 2006a; Scholz, 2006b). This development is also part of the Transformation of Rural and Urban Spatial Structure (TRUST) project, and involves the installation of a number of swales, underground storage tanks and retention ponds to reduce stress on the combined sewer system (Scholz *et al.*, 2006a). New regulations have been enforced, including limits on the granting of planning permission to developments which ensure that no additional runoff will impact the existing sewer system during a storm event (Scholz *et al.*, 2006a). For the Belvedere Hospital area, an interconnected network of swales has been identified as the most appropriate SUDS option, with a detention pond also being created; reducing flood risk whilst providing recreational opportunities and improving the areas aesthetics (Scholz *et al.*, 2006a). The SUDS scheme also involves the use of permeable and porous pavements in car parks and feeder roads to increase flood water retention capacity (Scholz *et al.*, 2006a). For the Celtic FC and Pollok Centre areas, integrated underground storage systems have been proposed in addition to infiltration networks which will deliver runoff from surfaces such as roofs and pavements to the underground unit and storage tank (Scholz *et al.*, 2006a; Scholz *et al.*, 2006b). As far as the economics of these SUDS options in Glasgow are concerned, a cost-benefit analysis reveals the initial investment costs for these SUDS solutions is comparable to those for a traditional drainage system (Scholz *et al.*, 2006a), although maintenance costs for SUDS are on average 30% lower (Butler and Davis, 2000; Broad and Barbarito, 2004).

A number of 'Green Streets' programs in the US employ SUDS techniques to adapt cities to climate change. For example, Philadelphia's Green Streets Program comprises of multiple sustainable drainage measures including infiltration trenches, vegetated swales, pervious pavement projects, and constructed wetlands for stormwater (Berkooz, 2011). Green streets in the city are very effective, being able to drain an area of 7.5 million square feet; intercepting and storing water from streets and pavements before it infiltrates into the soil and recharges groundwater, thus reducing the likelihood of urban flooding (Berkooz, 2011).

3.5.8 Building measures

A wide variety of building measures are being implemented across Europe for climate change. These include adaptation measures aiming to reduce the effects of climate change in buildings by utilising measures such as insulation, air conditioning and passive ventilation systems; and also those to mitigate future climate change, such as improvements in energy efficiency, low energy buildings and public transport. These two purposes of building measure will now be discussed in detail, although there is some overlap.

A number of alterations can be made in the building design process to adapt urban areas to climate change. These include the use of passive ventilation measures, building orientation, and shading to reduce the risk of overheating.

<u>Natural ventilation</u>

Natural or passive ventilation are measures to reduce indoor temperatures and increase thermal comfort of buildings have been employed in a number of European projects. Passive night cooling has large potential in parts of Europe, reducing temperatures inside buildings overnight, which allows them to provide a heat sink, and absorb heat gains during the daytime, thus removing or reducing the need for mechanical cooling (Artmann *et al.*, 2008). Passive night cooling was found to reduce inside building summer temperatures by 20-50% in Helsinki, Copenhagen, Potsdam, Oxford, Paris, and Zurich (Artmann *et al.*, 2008). However, in cities with warmer climates such as Madrid and Athens, the effects of passive night cooling are simulated to become negligible during summer under future climate conditions, effective instead during the spring and autumn seasons where this technique could reduce temperatures by 20-55% (Artmann *et al.*, 2008). These findings suggest that in southern Europe, alternative passive cooling techniques, such as evaporative cooling, may be required in the future to adapt buildings to climate change (Artmann *et al.*, 2008). In contrast, passive cooling alone is likely to continue to have significant potential over a minimum period of the next few decades in northern and central Europe in cities such as Copenhagen and Helsinki (Artmann *et al.*, 2008).

A number of existing developments have included passive ventilation as part of their design. For example the low energy residential settlement in Borgo Solare, Italy (Aste *et al.*, 2010) and the Open University design studio in Milton Keynes, UK (Zimmerman and Anderson, 1998). The Beddington Zero Energy Development also utilises a passive ventilation system which includes the installation of cowls on the roofs to draw wind into the building; making use of pressure and temperature differences to increase thermal comfort (Chance, 2009). Windcatchers operating a wind and buoyancy driven split-duct roof in a new secondary school building in London are another passive ventilation measure (Mavrogianni and Mumovic, 2010). Research found these to be an effective measure in the short term, able to alleviate the problem of overheating until the 2020s (Mavrogianni and Mumovic, 2010). However, as daytime temperatures increase under climate change to reach 30-35°C in the 2050s, higher airflow rates will be required to keep daytime indoor temperatures below 28°C, and the windcatchers will be inadequate by the 2080s (Mavrogianni and Mumovic, 2010).

Other forms of passive ventilation include the use wind towers and cross-ventilation to aid air circulation and heat exchange in buildings (e.g. Zimmerman and Anderson, 1998). Night-time cooling, utilising natural ventilation to remove heat which has accumulated inside buildings throughout the day, is a measure used both at the IONICA headquarters building in Cambridge, and in a single-family residence in Vila Nova de Gaia, Portugal (Zimmerman and Anderson, 1998). In these examples, night cooling, in combination with other building design measures eliminate previous requirements for mechanical cooling (Zimmerman and Anderson, 1998).

<u>Mechanical and passive cooling</u>

In a UK mixed-mode building, a mechanical ventilation system has been installed which supplies air through a floor void and extracts it through light fittings (Holmes and Hacker, 2007). The system includes an adiabatic spray to cool the return air stream and is able to control window ventilation, so that if the building's internal temperature exceeds 25°C, the windows close to automatically trigger the mechanical ventilation system (Holmes and Hacker, 2007).

Phase change materials (PCM) are yet another technology that can be used in building design to reduce the impact of climate change on urban buildings, although the suitability of this as an adaptation measure is highly dependent on climate. The installation of PCMs and a ceiling ventilation system shows much potential in the Italian cities of Milan, Rome and Crotone, being able to reduce peak cooling loads and maintain thermal comfort in the rooms even on hot days (Corgnati *et al.*, 2007). Research has shown that in locations such as Paris, Madrid and Athens, a 26°C wallboard can provide a considerable amount of passive cooling, for example offsetting room temperature in Athens by a maximum of 3°C and reducing overheating hours by 18% (Colclough *et al.*, 2009). It appears that the ability of a wallboard to provide cooling is dependent on climate, and hence a 26°C wallboard is ineffective in the temperate location of Belfast, where it has potential for adverse effects; absorbing heat from the heating system, and releasing it when not required (Colclough *et al.*, 2009). This highlights the need for care to be taken when selecting the most appropriate phase change temperature for a given location to ensure that the installation of PCM wallboards does not result in maladaptation (Colclough *et al.*, 2009).

Concrete slab cooling is another measure used in a number of buildings such as the DOW building headquarters, and Sarinaport office building in Fribourg, Switzerland (Zimmerman and Anderson, 1998). This provides both heating and cooling functions, with the concrete slabs being able to store excess heat, releasing it once the room cools; or to absorb energy from the surroundings when temperatures are too high (Zimmerman and Anderson, 1998). This system utilises heating ventilation and air-conditioning technology, allowing the thermal load accumulated during the day to be released at night via air coolers, with low energy consumption, maintenance costs, and operational savings (Zimmerman and Anderson, 1998).

Heating and cooling can also be provided by alternative sinks such as the ground and aquifers. A ground heating and cooling system has been installed at the Schwerzenbacherhof Office and Industrial Building, Switzerland (Zimmerman and Anderson, 1998). This passive heating ventilation and air-conditioning system works by pre-heating ventilation air underneath the building during winter, and cooling incoming air during the summer; able to reduce peak-demand for cooling, with a high-peak load performance in both seasons (Zimmerman and Anderson, 1998).

The Groene Hart Hopsital in Gouda, the Netherlands, and the SAS Frösundavik office building in Stockholm utilise aquifers as a thermal store (Zimmerman and Anderson, 1998). Over the summer, the aquifers accumulate heat, which is used in the winter for heating purposes; and over the winter, they store cool, which can be used to cool buildings during summer via a series of cold and warm wells (Zimmerman and Anderson, 1998). In addition to improving thermal comfort, this system leads to reduced GHG emissions, with an energy use reduction of 65% in the SAS office building compared to that of a conventional system (Zimmerman and Anderson, 1998).

Hybrid Adaptable Thermal Storage (HATS) systems and materials were also found in this review to adapt buildings to climate change. These are being simulated in the Netherlands for residential dwellings of the Zonne-Entree project in Apeldoorn (Hoes *et al.*, 2011). Model results show that HATS systems here have the maximum capacity to reduce summer over- and under- heating hours by 1,295% compared to a conventional permanent low thermal mass concept (Hoes *et al.*, 2011).

Shading and daylighting

This review found a number of buildings in the UK which utilise a mixture of shading and daylighting to reduce building energy requirements (e.g. Holmes and Hacker, 2007). These include a new secondary school in London which has maximised on daylight hours with the aspect of many rooms being south-facing (Mavrogianni and Mumovic, 2010). The Lanchester Library of Coventry University also uses daylighting via four lightwells, reducing electricity requirements for artificial lighting by maximising solar gains (Krausse *et al.*, 2007). A combination of natural ventilation, daylighting and passive cooling in this building are able to maintain the interior temperature at a maximum of 5°C below the ambient temperature during the summer season and have also reduced building energy use for electricity and gas by 51% compared to a standard air-conditioned building (Krausse *et al.*, 2007).

A numerical case study of a building in Slovenia examines the effect of a green building design (see Leskovar and Premov, 2012a). This hypothetical building design uses large glazing areas to maximise sunlight penetration into the building, with the south-oriented façade having a glazing-to-wall ratio of 27.6% (Leskovar and Premov, 2012a; b). Large glazing areas in the south-oriented exterior walls of these buildings reduce demand for heating and improve energy efficiency; however, in contrast, increasing the glazing surface in the north-oriented external wall had adverse effects being found to increase energy demand for heating (Leskovar and Premov, 2012a). The effects of shading are also modelled, with the top floor at the south side of the building designed to have an overhang to block direct solar radiation during the summer, reducing demand for cooling; and conversely allowing this radiation to enter during winter months when the sun is lower in the sky (Leskovar and Premov, 2012a).

Window shading and inclined roofs were found to have potential for climate change adaptation in Cyprus (Florides *et al.*, 2000). Of these, an inclined roof was identified as having the most potential, able to reduce summer air-conditioning cooling loads by 41-55%, whereas window shading was much less effective, only able to achieve a reduction of 8-20% (Florides *et al.*, 2000).

3.6 Water

To help the water sector and to relate synergies and integration to environmental, economic, urban and social sectors, the EU policies of the Water Framework Directive and the Water Scarcity and Drought Communication have been developed. These evaluate the supplies and demands for water, as well as the overall state of the water environment. Quevauviller (2011) states that climate change is not seen as an anthropogenic pressure to the Water Framework Directive, yet over many decades, scientists recognise that climate change does cause changes to water resources to many sectors and their impacts. Climate change and mitigation can influence many steps of the Water Framework Directive, and can exacerbate existing problems. A European White Paper in adapting to climate change helps identify these, and then considers what adaptation strategies can increase resilience over a wide range of sectors influenced by water management, working within the remit of other frameworks and directives (e.g. the EU Floods Directive).

Adaptation of water resources relates to the supply and demand, and the efficiency of the delivery of water between them. There are many adaptation interventions that can be carried out to reduce the impacts of climate change on water resources, including flood risk (Dawson *et al.*, 2011). These can be separated into five key areas:

3.6.1 Increased infiltration

Methods of increasing infiltration include changing tillage practices (e.g. Gordon *et al.*, 2011; Tomer and Schilling, 2009), extensification of farming practices and storm water source control. Further details are given in Section 3.1 of this report.

3.6.2 Increased storage

Reduced runoff can be achieved by reducing field drainage to improve localised water storage (Wilson *et al.*, 2011), afforestation to increase evapotranspiration (see Section 3.4.1, and Ortigosa and García-Ruiz, 1995; Robinson *et al.*, 2003; Trabucco *et al.*, 2008), retaining water through detention ponds, rainwater harvesting (Section 3.5.5) and restoration of wetlands (Section 3.2.5) (Mwenge Kahinda *et al.*, 2010; Glendenning and Vervoort, 2011; Wilson *et al.*, 2011). The restoration of river channels and the surrounding floodplain can also increase temporary storage area (Rohde *et al.*, 2006; Buijs, 2009).

3.6.3 Reduced flow rate

Reducing peak flow rate can reduce the effects of river flooding; this can be done by altering the main drainage channels of a river or, in urban areas, re-opening culverted watercourses (see Section 3.5.7).

3.6.4 Reduced flood impact

Reducing impact of floods include defences (Section 3.5.6), land use planning - making space for the river and flood water - such as flood plain restoration (Section 3.2.5, and see also Klijn *et al.*, 2004; Howgate and Kenyon, 2009). In some areas (e.g. England, Charlton and Arnell, 2011), climate change is expected to produce restrictions in water availability, and thus the demand for water also needs to be managed.

3.6.5 Demand management

Changes to demand include upgrading infrastructure, fitting water efficient equipment, promoting efficient use through education, water resource and recycling, including rainwater and more efficient tariffs (see Section 3.5.5 and Arnell and Delaney, 2006). This may be governed to include local (e.g. hosepipe bans), national (e.g. campaigns to use less water) and EU policies (directives in the appropriate use of water, balancing ecosystem needs).

The first four of these have important implications for other sectors and so they are covered in cross-sectoral interactions (Section 7.2.2), but demand management has little direct impact on the sectors under consideration and thus it is not discussed further here, although some sectors do mention that an adaptation may reduce demand, e.g. in agriculture.

4. Mitigation options by sector

4.1 Agriculture

4.1.1 Agriculture in Europe

Nitrogen budgets and biological nitrogen fixation

The literature search found a number of articles estimating nitrogen budgets for farmland. For example, de Vries *et al.* (2011) modelled land nitrogen (N) budgets for the EU, finding total N inputs to European agriculture in the range of 23.3-25.7 M t N y^{-1}. In contrast, N uptake was much more variable, at 11.3-15.4 M t N y^{-1}, and hence total N surpluses of 10.4-13.2 M t N y^{-1} were calculated. In a similar modelling study, Leip *et al.* (2011) found N surpluses in Europe of 55 kg N ha^{-1} y^{-1} (soil budget); 65 kg N_2O-N ha^{-1} y^{-1} (land budget); and 67 kg N ha^{-1} y^{-1} (farm budget). Model results revealed farms in Romania and Bulgaria to have the highest farm nitrogen use efficiency (estimated at around 50%), due to the dominance of extensive agriculture and crop production; whereas farms in Ireland and Slovenia had the lowest farm nitrogen use efficiency values of around 15%, as these systems are highly specialised around livestock products (Leip *et al.*, 2011).

As far as actions to reduce N losses, and increase nitrogen use efficiency are concerned, crops such as legumes have the potential to fix nitrogen by themselves and hence have the potential to mitigate climate change (Brehmer et al, 2008; James and Baldani *et al.*, 2012; Jensen *et al.*, 2012). The amount of nitrogen fixed by the worlds grass and legume pastures is estimated to be between 13-682 kg N $ha^{-1}y^{-1}$, with around 26% of this stored below ground by the decomposition of legume roots and nodules (Ledgard and Steele, 1992). On a global scale, soybean crops fix a particularly large amount of nitrogen, contributing 77% of the N fixed globally by legumes (Herridge *et al.*, 2008). On a smaller scale, the biological nitrogen fixation (BNF) by white clover in upland and marginal areas of the UK is estimated at 100-150 kg N ha^{-1} y^{-1} (Newbould *et al.*, 1982). Fixing higher amounts of N into the soils in these areas provides a sustainable means to improve pastures, whilst being an economically attractive option for making agriculture less marginal in these areas (Bohlool *et al.*, 1992; Newbould *et al.*, 1982). To illustrate this point, it has been estimated that establishing existing varieties of white clover and strains of rhizobium in the UK would improve an additional 10% of the better upland soils and fix approximately 50 k t N y^{-1}; worth £20 million at 1982 fertiliser prices (£0.40 kg^{-1}) (Newbould *et al.*, 1982). Additional benefits of BNF are well documented in the literature; supplying nitrogen to current and succeeding crops resulting in less demand for mineral fertilisers and sizeable reductions in NO_3 leaching (Bohlool *et al.*, 1992; Danso *et al.*, 1992; Giller & Cadisch, 1995; Bøckman, 1997; Brehmer *et al.*, 2008). Secondary consequences of this reduced demand for fertiliser include a reduction in farm inputs and fossil fuel use; both impacting on GHG emissions (Bohlool *et al.*, 1992; Jensen *et al.*, 2012). The net effect of farming legumes for climate change mitigation is somewhat complex (Jensen *et al.*, 2012). On one hand, this results in large emissions savings from fertilisers, these being carbon intensive with approximately 300 Tg CO_2 released each year from the production of 100 Tg nitrogen fertiliser (Jensen *et al.*, 2012). However, in contrast to these savings, large amounts of CO_2 are respired annually by the root nodules of agricultural legumes, estimated to be around 350-500 Tg CO_2 as a result of the biological fixation of 33-46 Tg per year (Jensen *et al.*, 2012). Despite this, it is important to note that the CO_2 resulting from respiration, although substantial, does not represent a net contribution to atmospheric CO_2 concentrations, whereas that released from fossil fuels in the manufacturing of N fertiliser does (Jensen *et al.*, 2012).

Tree legumes are also able to fix N biologically, and hence it is possible to store N in agroforestry, as well as traditional cropping systems (Danso *et al.*, 1992; Peoples *et al.*, 1995). Legume trees planted for the purpose of BNF are associated with a number of synergies. Firstly, they benefit the soil by restoring fertility, reducing degradation and erosion, whilst also providing high quality forage, in addition to providing a resource through food and timber (Danso *et al.*, 1992; Peoples *et al.*, 1995). In terms of mitigation potential, the most effective tree species in fixing nitrogen are those such as *Leucaena leucocephala* and *Sesbania rostrata*; able to store upwards of 500 kg N ha^{-1} y^{-1}, whereas in contrast, trees such as *Sesbania sesban* store substantially less N (often by more than an order of magnitude) (Danso *et al.*, 1992). As a result, it can be difficult in some cases to detect the net effect of the trees on total soil N (Peoples *et al.*, 1995). Finally, it is important to note that the effectiveness of agroforestry systems to fix N is highly site-specific, influenced by a number of factors including for example tree age, soil moisture, temperature, soil nitrogen levels and plant nutrient deficiencies (Peoples *et al.*, 1995).

A recent study examined the potential of legume trees grown on marginal land in terms of nitrogen fixation and as a bioenergy crop (Biswas *et al.*, 2011). The authors highlighted a number of benefits; the trees being able to grow in N and P poor soils, whilst producing resources, such as seeds which are often used as feed and fodder. *Pongamia* was found to be the species with the most BNF potential, however, research to develop superior varieties and cultivars is still in its infancy, and other types need to be considered for use in regions such as Europe (Biswas *et al.*, 2011). In contrast to this, a study conducted by Brehmer *et al.* (2008) concluded that legumes were not sustainable when utilised as a bioenergy crop alone, instead, the additional benefits discussed above should be taken into account.

Despite the potential described above for legumes to act as a form of climate change mitigation, this has rarely been considered in the literature (Smith *et al.*, 2008), although a number of studies do examine methods which can be used to improve N fixation. Firstly, research has shown that altering the management of legumes has the potential to increase their potential for BNF (Cowling, 1982; van Kessel and Hartley, 2000). For example, the BNF by agricultural grasslands in the UK could be improved if white clover in swards was cultured more carefully, and by an increase of leguminous forage crops to provide a sustainable feed (Cowling, 1982). Furthermore, improvements in the general growing conditions for grain legume crops (e.g. by improving pest management practice, improving soil structure, and reduced tillage) have been shown to result in crops having a heightened demand for N, and therefore able to fix more N_2 (van Kessel and Hartley, 2000). Alternatively, selecting rhizobium-host plants which are less sensitive to mineral nitrogen can also improve N_2 fixation. This method does, however, carry with it number of antagonisms, with the nitrogenase enzyme becoming less sensitive to available N, resulting in accumulations of N at the end of the growing season, and hence an increased potential for N losses from the system through leaching and denitrification (van Kessel and Hartley, 2000).

There appear to be inconsistencies between studies with regards to soil N accumulation as a result of biological fixation. For example, Ledgard and Steele (1992) found that in the short-term, soil inorganic N content increased during dry periods, and where nitrogen fertiliser was utilised, however in the long-term, BNF did cause nitrogen to accumulate in the soil. In addition, it appears there is a disparity between the results from field and modelling studies, with experiments indicating a potential for N_2 fixation in the range of 200–400 kg N ha^{-1} y^{-1} for a wide range of legumes, whereas field studies find N_2 fixation to be substantially lower at 0-200 kg N ha^{-1} y^{-1}, suggesting a much lower potential due to nutrition limitations, drought, pests or disease (Herridge *et al.*, 2008). The ability of different crops to fix nitrogen appears to be highly variable. For example, a study by Walley *et al.* (2007) examined the nitrogen fixing potential of pulse crops in the US Great Plains, reporting some varieties to fix relatively high levels of N_2, resulting in net soil N accretion (e.g. faba bean, field pea, lentil); whereas others resulted in no net, or even a negative change in soil N content (e.g. desi, kabuli chickpea, common bean).

Finally, it is important to note that a number of papers discuss the possibility of engineering crops, such as cereals to enable them to fix nitrogen themselves, hence sustaining their own growth and yields whilst reducing atmospheric nitrogen concentrations (Kennedy *et al.*, 1997; Thomson *et al.*, 2012). However, this remains in the research stages, and the review found no evidence of this being currently implemented in Europe.

Fertilisers

The mitigation potential of fertilisers was discussed in numerous studies found in the literature search. These include altering the timing of fertiliser treatments, as well as their amount and application method. This will be discussed in more detail, although it is important to note that the GHG abatement potential from fertilisers is highly variable, influenced by fertiliser-type, and application rate, in addition to soil type and climatic conditions (Hillier *et al.*, 2012).

<u>The impact of fertiliser type</u>

Changing fertiliser practices is discussed by a number of authors as a potential mitigation action. Hillier *et al.* (2012), for example, identify switching from old to new fertilisers as a form of mitigation: able to reduce fertiliser-induced soil N_2O emissions by around 20%. In another study, the replacement of synthetic fertilisers with mineral and organic fertilisers (such as manure) in south-east Italy was found to increase soil fertility and long-term soil carbon storage (Triberti *et al.*, 2008). Alternatively, liming applications to farmland have been identified as another effective mitigation strategy (e.g. Fornara *et al.*, 2011), as they are associated with a reduction in the use of nitrogen fertiliser applications in the UK. The application of solid manure is already a mandatory practice in Denmark, with solid pig and cattle manure estimated to reduce emissions by 226 and 101 kg CO_2-eq ha^{-1}, respectively (Hansen *et al.*, 2006).

<u>Timing of fertiliser applications</u>

Altering the timing of fertiliser treatments is shown to take two forms: (a) the splitting of applications; and (b) limiting application to favourable climatic conditions. Firstly, as far as splitting is concerned, a study conducted in France which split the application of nitrogen fertiliser (one application in early March, and the second in mid-April) found that this resulted in a higher fertiliser nitrogen use efficiency, in addition to a higher growth potential for wheat in early spring (Durandeau *et al.,* 2010). This apparent benefit for yields associated with split fertiliser applications was quantified in another study as an increase of 6% (del Grosso *et al.*, 2009). Secondly, concerning point (b), a review by Luo *et al.* (2007) found N_2O emissions following fertiliser applications to be highest in wet soils and, hence, limiting applications during wet months with saturated soils and slow growth can be effective in reducing N_2O emissions from grazed pastures (Luo *et al.*, 2010). In addition, postponing manure applications until after grazing has also been shown to reduce N_2O emissions as a result of less surplus mineral nitrogen being present in the soils and, hence, reducing nitrate leaching (Luo *et al.*, 2010).

<u>Fertiliser application methods</u>

Weiske *et al.* (2006) compared the effectiveness of various manure application techniques for reducing GHG emissions in Europe. The results showed that manure application by trail horse and injection reduced farm GHG emissions on average by 0.7% and 3.2% respectively compared to broadcasting (Weiske *et al.*, 2006). Research has shown manure injection and the incorporation of manure into the soils after spreading to be some of the most effective measures in reducing ammonia emissions in Europe, with additional reductions in N_2O emissions (Malgeryd, 1999; Brink *et al.*, 2001). In addition, the irrigation of soils after spreading has been shown to delay and reduce ammonia emissions by preventing the manure from drying out, as well as transporting NH_{4+} and NH_3 into the soil (Malgeryd, 1999).

Fertiliser application methods which take into account individual site requirements were also identified in the literature search as effective means to reduce emissions and lateral losses of fertilisers from the field. This can be done using a handheld NDVI sensor, with a study by Oritz *et al.* (2008) describing how this would reduce unnecessary nitrogen fertiliser inputs, hence, increasing resource use efficiency and providing economic savings. Overall, the study found site-specific application assessments reduced trace gas emissions, although this was not discussed explicitly as a mitigation action (Ortiz *et al.*, 2008). Practices such as this, which adjust the amount of nitrogen fertiliser and manure applied to the field to meet crop N demands, have been shown to be most effective in farming systems with large nitrogen surpluses (Kros *et al.*, 2010). Results showed this to have the largest mitigation potential for the European Union, predicted to be able to reduce N_2O emissions by 12% (Kros *et al.*, 2010).

In the US, a study by Genskow (2012) assessed the impact of nutrient management plans (NMPs) for farmers in Wisconsin. These appear to be relatively successful, with over 80% of farmers following plans on the majority of their farmland. As a result of the scheme, 47% of participants in the study decreased nitrogen fertiliser applications on average by 84 kg N ha^{-1}; however 51% increased fertiliser N applications by an average of 89 kg N ha^{-1}. As far as phosphorus pentoxide (P_2O_5) emissions are concerned, NMPs saw 46% of farmers decrease their application rates, whereas 47% increased their application rates. Overall, the plans saw 65% of participants reduce their use of commercial nitrogen, and 51% decrease their use of commercial phosphorous on the farm. Secondary impacts of the NMPs discussed in the study were either a moderate (53%) or major (38%) improvement in surface and groundwater quality, without any negative effect on yield (Genskow, 2012). It may be possible to introduce such plans in Europe, although no studies found in the review discussed this potential.

<u>Extensification</u>

As far as the amount of fertiliser applied to farmland is concerned, Gregorich *et al.* (2005) studied the GHG impact of varying the amount of mineral nitrogen fertiliser applied to the soils. The authors identified a linear relationship between soil N_2O emission and the amount of mineral nitrogen fertiliser applied (Gregorich *et al.*, 2005). Similarly, in Western Europe, it was found that halving the rate of nitrogen inputs resulted in a 27% reduction in the net GHG balance (Lehuger *et al.*, 2011). Decreasing the amount of fertiliser applied can have a large mitigation potential, with a long-term example of the effectiveness of this option given by Leifeld and Fuhrer (2005). The authors studied agricultural trends in Switzerland since 1990, finding that mineral nitrogen fertiliser applications in Switzerland have decreased by 16.5% since this time, with a subsequent reduction in N_2O emissions from this sector.

Reductions in the amount of fertiliser applied in farm systems also reduces emissions indirectly through reduced fuel consumption, as fewer hectares of pasture are cut and bailed for hay, and also less synthetic fertiliser is applied (Stewart *et al.*, 2009). A study conducted in Denmark found that a 41% reduction in nitrogen fertiliser application resulted in a 39% decrease in the amount of energy incorporated in fertilisers, pesticides and lime; in addition to a 12% reduction in total energy based emissions (Bennetzen *et al.*, 2012).

Despite the effectiveness of reductions in fertiliser application in reducing emissions, it is important to recognize also the link between fertiliser treatments and the productivity of a farm system. In the afore mentioned studies, although a reduction in the quantity of fertiliser applied was shown to reduce emissions, whether that be directly or indirectly, it can also suppress yields (Stewart *et al.*, 2009). In a cattle farming system where fertiliser application was reduced, the decline in productivity effectively increased the amount of land needed to meet the same requirements, i.e. the area on which the cattle grazed needed to be expanded, resulting in fewer hectares free for other crops with a saleable value (Stewart *et al.*, 2009). Hence, a reduction in fertiliser use was linked to a large increase in emissions per tonne of protein (Stewart *et al.*, 2009). It is therefore inherently important to examine the effect of a given mitigation practice not only on net GHG emissions, but also on yield, as a practice which reduces emissions but decreases productivity will mean that in the long-term, more land will need to be converted to agriculture to deliver the same yield and, hence, overall GHG emissions increase. This point was illustrated by Del Grosso *et al.* (2009) who used the DAYCENT biogeochemical model to simulate the effect of reduced fertiliser application. The study accounted for a number of variables including soil class, finding that although sites with reduced quantities of fertiliser applied experienced lower nitrogen losses, crop yields were also reduced by a similar proportion (Del Grosso *et al.*, 2009).

Intensification

In contrast to reducing fertiliser applications, a number of studies from this review considered increasing nitrogen input to the farm-system as a form of mitigation. For example, Meyer-Aurich *et al.* (2012) discussed how this approach in Germany and Denmark would increase yields and reduce the land area needed for crop production. As less land area would be required to produce the same amount of crop, it would then be possible to use the surplus land for energy crops (Meyer-Aurich *et al.*, 2012). However, the authors highlight that this would only be justified if GHG mitigation with the additional land is greater than 9-15 t CO_2 eq ha^{-1}, and as the mitigation potential of bioenergy production from energy crops is not often in this range, it would only be justified in exceptional cases to mitigate GHG emissions using bioenergy (Meyer-Aurich *et al.*, 2012). In addition to increased productivity, a modelling study for Spain found that as the rate of N fertiliser application was increased, SOC stocks also increased, with applications of 0; 60; and 129 kg N ha^{-1} resulting in sequestration rates of 30.6 ; 33.5; 35.8 Ma C ha^{-1}, respectively (Alvaro-Fuentes *et al.*, 2012).

Despite these apparent benefits, Lehuger *et al.* (2011) found the effect of increased fertiliser application on productivity to be limited, with a 50% increase in mineral N fertiliser input increasing net primary productivity by only 1%. In addition, this practice was associated with a number of antagonisms, for example a GHG balance 22% higher than the control, 17% higher N_2O emissions, and 27% higher indirect emissions.

Manure

This search identified a few studies discussing changes to manure practices as a form of climate change mitigation, discussing the potential of manure to sequester carbon and also the effect of various manure handling, storage and application methods on GHG emissions.

Carbon sequestration

Animal manure contains a high percentage of carbon (40-60% on a dry weight basis) and, hence, applications of manure to the soils has the potential to increase their carbon content (CAST, 1992). The carbon sequestration potential of manure from land applications has been reviewed and it has been found in numerous studies that SOC sequestration in terms of land area displays a positive relationship with the rate of manure application (e.g. Sommerfeldt *et al.*, 1988; Gupta *et al.*, 1992). However, it is important to note that few of the studies reviewed examine the effect of increased manure application rates using data from the whole-farm system (Franzluebbers, 2005).

Much work has been conducted by Smith *et al.* on the impact of different rates of manure application on farms (e.g. Smith *et al.*, 1997; 2000b; 2001). One such study examined the carbon mitigation potential of two contrasting rates of application of animal manure, those being 6.1 t ha^{-1} y^{-1} and 20 t ha^{-1} y^{-1} (Smith *et al.*, 2000a). It was found that the lower application rate had a maximum carbon mitigation potential of 11.10 Tg y^{-1}, and could be applied to a maximum possible 86.6% of Europe's arable land area (Smith *et al.*, 2000a). The higher application rate of 20 t ha^{-1} y^{-1} offered a higher carbon mitigation potential of 13.42 Tg y^{-1}, but was fairly limited in terms of the maximum arable land area it could be applied to in Europe (only 26.5%) (Smith *et al.*, 2000a). In a contrasting study, Vleeshouwers and Verhagen (2002) estimated that if applied to the total 231 M ha of arable land in Europe, an application of 10 t ha^{-1} manure would mitigate 350 Tg C y^{-1} for the first four years after implementation, however, this potential was found to be highly spatially variable. For example, carbon sequestration rates associated with manure applications were highest in south-western and south-eastern areas of the continent, (i.e. Spain and Turkey) where dry conditions during the summer season and a low soil C content reduce the rate of decomposition of SOM (Vleeshouwers and Verhagen, 2002). It was found that in eastern Europe, more carbon mitigation potential was provided by annual applications of 10 t of farmyard manure per hectare; whereas in western Europe, more carbon would be sequestered from the conversion of arable land into grassland (Vleeshouwers and Verhagen, 2002).

On a smaller, UK, scale, Smith *et al.*(2000c) found the maximum carbon potential for a 20 t ha^{-1} application of manure to be 1.75 Tg C y^{-1} when applied to a possible 45.3% of the arable land area. Although this appears to be relatively low, the authors do note that if implemented as part of a combined management strategy, the scheme would have a higher mitigation potential than that quoted above (Smith *et al.*, 2000b).

As far as the timeframe associated with this mitigation action is concerned, modelling results from a study in northeast Italy show that this potential is greatest in only the short-term, with up to 62 g C m^{-2} sequestered in the first 5 years, becoming less efficient in the long-term compared to practices such as reduced tillage (Lugato and Berti, 2008). Similarly, research by Smith *et al.* (1997) found that an amendment rate of 10 t ha^{-1} animal manure applied to all European agricultural soils would increase the total SOC by only 5.5% over the next century, amounting annually to carbon storage of only 12.58 Tg y^{-1}, or 1.37% of annual anthropogenic emissions in western Europe (Smith *et al.*, 1997). This mitigation option therefore has limited potential to increase soil carbon stocks over the mid-term (Smith *et al.*, 1997).

There remain some discrepancies between studies concerning the carbon sequestration potential from manure amendments to soils. For example, a review paper by Powlson *et al.* (2011) found that for the majority of cases the observed increase in SOC associated with amendments of animal manure to agricultural land did not create an additional transfer of carbon from the atmosphere to the soils (Powlson *et al.*, 2011). Furthermore, if farms adopt this practice, increased amounts of manure will need to be transported, having a secondary impact on the environment. It has been estimated that as a result of increased demand for fuel, the release of particulates from fuel combustion, and increased trace gas emissions from the transport sector negates around 30% of the benefit from carbon sequestered if the average transport distance is taken to be 100 km (Smith and Smith, 2000; ECCP, 2001). Despite these emissions associated with manure applications, Lehuger *et al.* (2011) found that the cessation of manure amendment to soils in western European farming systems, although leading to reductions in N_2O emissions, was the worst option for mitigation. The cessation of manure applications were shown to result in a loss of carbon input for the entire crop rotation, not compensated for by the former reduction in N_2O emissions (Lehuger *et al.*, 2011). As a result, the study found that stopping manure applications would result in a 45% higher GHG balance overall, but a 20% reduction in N_2O emissions (Lehuger *et al.*, 2011).

Manure management

Manure management prior to field applications has the potential to alter GHG emissions associated with manure amendments to soils. For example, Sommer *et al.* (2009) used a livestock model for farms in Sweden, Denmark, France and Italy to examine the impact of separating slurry into a solid and liquid fraction in farms which utilise slurry-based manure management. Model results showed that changes in manure management were able to significantly affect emissions of CH_4 and N_2O, as well as carbon storage in the soils (Sommer *et al.*, 2009). It was also found that effect of this practice could vary significantly depending on livestock farming technique and climatic conditions, both of which must be taken into consideration when examining changing management practices for the purposes of mitigation at a given location (Sommer *et al.*, 2009).

Altering the composition of manure before field applications also has the potential to reduce phosphorous (P) losses (O'Rourke *et al.*, 2012). This can be achieved if either the P concentration of the manure is decreased, or alternatively if the soluble P fraction is reduced by amending the manure with a material which sorbs phosphorous before it is applied to the soils (e.g. co-blending with manure with a water treatment residual, WTR) (O'Rourke *et al.*, 2012). The results for the study from Ireland showed P concentrations in run-off were reduced by half in only two months (3.7 mg P L^{-1} with WTR compared to 7.6 mg P L^{-1} for the control) (O'Rourke *et al.*, 2010). Despite this reduction in P losses from the farm system, this is not yet a viable management practice in the study area, as the spreading of WTRs in Ireland is not legalized and therefore the authors advise that it may be appropriate to examine how P losses from existing manure management practices can be reduced.

In contrast to studies which examine only the impacts of field applications of manure, a range of handling and storage strategies can alter the GHG emissions associated with manure, e.g. increasing the frequency at which manure is removed from animal housing (Massé et al., 2008). In a similar study conducted for Europe, it was found that the frequent removal of manure from housing to outside storage reduced farm GHG emissions by up to 7.1% (Weiske *et al.*, 2006). Results showed that although the daily removal of manure from animal housing had substantial mitigation potential when considered alone, being able to decrease emissions from animal houses by 97%; when examining a whole farm system, this potential was substantially reduced as a result of large increases in emissions from manure stores (Weiske *et al.*, 2006). A modelling study by Sommer *et al.* (2009) found similar results, with a reduction in the time duration of in-house manure storage able to reduce trace gas emissions by 0-40% (Sommer *et al.*, 2009). This is a sizeable reduction, and in addition to manure removal, leaving cows outside during summer nights and reducing the depth of residual manure left in the tank after land application are able to reduce emissions by an average of 12%; and CH_4 emissions by 24%, respectively (Massé et al., 2008).

The temperature at which manure is stored is another factor known to impact GHG emissions from stores. Dalgaard *et al.* (2011) estimate that CH_4 emissions from livestock houses and manure stores in Denmark could be reduced by almost a third if the temperature of slurry channels was reduced to 10°C. Although this potential exists, it is not cost effective to maintain slurry channels at a temperature below 15°C, and hence CH_4 emissions would be reduced by only 18%.

When examining the potential of these options to mitigate emissions, it is essential to take into account the importance of site-specific factors. As far as increasing the frequency of manure removal, for example, Masse *et al.* (2008) found that using a single emission factor for all farms in a region, as opposed to estimating CH_4 emissions on a site-by-site basis resulted in large errors.

Despite the benefits of reduced manure in terms of a climate change mitigation option, a number of studies discuss this primarily as a tool to improve water quality by reducing nitrogen losses from the farm system. One such example which does explore reductions in GHG emissions is Cherry *et al.* (2012), a study comprising of management plans in 34 farms in southwest England, where 65% of farmers followed a manure management plan, with measures such as avoiding autumn applications of slurry or poultry manure (Cherry *et al.*, 2012). Results showed a reduction in surplus nitrogen on the farms, leading to reduced leaching, and hence the authors suggest that improved management of manure should be a practice focused on for reducing N surpluses and GHG emissions from farms (Cherry *et al.*, 2012).

Livestock diet

A number of studies from this review discuss altering the diet of livestock as a way to reduce GHG emissions from farms (e.g. Bell *et al.*, 2011). Nahm (2007), for example, reviewed the mitigation potential of a number of modifications to livestock feeding programs, such as phase feeding, phytase and enzyme supplementation. These significantly affected both N and P emissions, with phase feeding reducing the amount of N and P excretion from chickens and pigs each from 10 to 33% and 10 to 13%, respectively. Authors such as Osada *et al.* (2011) have reported the impacts of a low protein diet supplemented with amino acids on nitrogen retention in swine manure. Experiments here showed that a low amino acid supplemented diet was able to reduce nitrogen excretion in the manure with no visible impact on animal growth, and a 39.1% reduction in global warming potential. In addition to dietary changes, improving the nutrient efficiency of feed is known to reduce pollutants in poultry and swine manure, with, for example, significant decreases in both nitrogen and phosphorous, as well as odour and a lower dry matter weight of the manure (Nahm, 2007). Feed amendment minerals, such as dietary zinc, can also be used to reduce ammonia emissions from livestock (Hunde *et al.*, 2012). As well as reducing emissions, this is associated with a number of synergies. Firstly, the volatilization of ammonia from poultry manure adversely affects animal welfare, hence, this improved; and secondly, the reduction of ammonia volatilization helped to protect human health and the environment (Hunde *et al.*, 2012).

One example of a specific mitigation action employed in Europe for livestock is that of anaerobic digestion technology in dairy, sow and pig farms in Finland (Kaparaju and Rintala, 2011). This was able to reduce GHG emissions due to the reduced production and use of fertiliser in combination with reductions from manure management. In addition, the manure was able to provide a renewable energy source, with up to 62.8 MWh of electricity per year from a farm producing 2000 m^{-3} of cow manure in a combined heat and power unit (Kaparaju and Rintala, 2011). As a result, it was concluded that the total GHG emissions that could be offset on the studied dairy cow farms were 177 Mg CO_2 eq y^{-1} (Kaparaju and Rintala, 2011).

Tillage

Conservation agriculture practices are another possible mitigation measure for the agricultural sector. These aim to minimise soil disturbance though methods such as reduced tillage and the cessation of ploughing (no-till). In addition, conservation agriculture focuses on introducing a permanent organic cover over the soil surface, whether that be from live cover crops, or organic residue, and also seeks to diversify cropping systems (Hobbs, 2007).

Although a substantial amount of the literature discusses the adoption of conservation agriculture practices, such as direct seeding (e.g. Munoz *et al.*, 2007; Khaledian *et al.*, 2010), minimum- (e.g. Giacomini *et al.*, 2010), reduced- (e.g. Akbolat *et al.*, 2008; Carlton *et al.*, 2012; Powlson *et al.*, 2012) and no- tillage (e.g. Smith *et al.*, 1998; Tebrügge and During, 1999; de Vita *et al.*, 2007; Stevens and Quinton, 2009; Blanco-Canqui *et al.*, 2012), this literature search found relatively few instances of work being conducted specifically for the purpose of climate change mitigation. Often the studies conducted for mitigation purposes were located outside of Europe, in regions such as the tropics, or took the form of review papers (providing no specific examples), and modelling studies analysing the potential of such practises in Europe under a number of future climate and economic scenarios. This may be because the tropics is the region with the greatest mitigation potential of conservation tillage and, hence, a greater number of studies have been conducted here (Paustian *et al.*, 1997).

For Europe, the adoption of conservation management practices appears to be related primarily to attempts to improve current agricultural conditions, for example, reductions in erosion rates or increases in soil fertility in semi-arid areas of the Mediterranean (Freibauer *et al.*, 2004; Alvaro-Fuentes *et al.*, 2008a). However, this search has found a number of articles discussing the use of conservation agriculture in Europe as a mitigation technique (e.g. Borin *et al.*, 1997; Six *et al.*, 2004).

Energy savings

In terms of indirect reductions in emissions, a number of studies note that conservation agriculture practices require less energy (e.g. Borin *et al.*, 1997; who calculated this to result in a 32% saving in energy per hectare) and, hence, a reduced need for the use of fossil fuels, such as diesel (Borin *et al.*, 1997; Khaledian *et al.*, 2010); both of which carry with them economic savings (Filipovic *et al.*, 2006). For example, Khaledian *et al.* (2010) calculated that direct seeding into mulch for corn and sorghum crops in France reduced energy inputs significantly (by as much as 18%), whilst conserving farm output (Khaledian *et al.*, 2010). Similarly, a no-tillage experiment in Croatia resulted in 87.8-88.1% reduced emissions from fuel consumption compared to conventional tilling (Filipovic *et al.*, 2006). It has been calculated that with complete adoption of no-tillage in Europe as much as 3.2 Tg C y^{-1} could be abated through reduced fossil fuel emissions from the agricultural sector (Smith *et al.*, 1998). Furthermore, a 100% conversion to no-till practices has the potential to cancel European fossil fuel agricultural carbon emissions, which, put into a global context, equates to around 0.8% of annual global anthropogenic CO_2-C emissions (Smith *et al.*, 1998). Other studies show that the scale of this carbon mitigation is relatively small, with few GHG savings (e.g. Powlson *et al.*, 2012). For example, Carlton *et al.* (2012) found the conversion from traditional to no-till practices in areas of the UK led to only a modest decrease in emissions at each site (<20%), with the extensive adoption of this management practice having the potential for a 15% reduction in future emissions. In contrast, compared to the effect on N_2O emissions of this practice, the mitigation potential from reduction in fuel use is often a magnitude lower (Antle *et al.*, 2012).

Carbon storage

As far as carbon is concerned, no-till systems allow organic carbon to accumulate, particularly in the upper soil as a result of the organic residues (Dersch and Böhm, 2001; Bescansa *et al.*, 2006; Alvaro-Fuentes *et al.* 2008a; Melero *et al.*, 2009; Kassam *et al.*, 2012). This is because no-tillage practices allow the CO_2 produced by the decomposition of soil organic matter (SOM) to diffuse more easily into the atmosphere, as it is produced closer to the soil surface. Studies vary on the magnitude of carbon sequestered in the upper soils as a result of no-tillage, with one study calculating a 310 ± 180 kg C ha^{-1} y^{-1} average annual increase in carbon of the top 0-30 cm soil layer when a traditional farm system was converted to no-till (Powlson *et al.*, 2012), and another finding that almost 60% of carbon mineralisation potential in no-tillage systems is located in the upper 5 cm of the soil profile (Oorts *et al.*, 2007). To put the carbon storage potential of contrasting management practices into perspective, a modelling study found the mean SOC stock in a Spanish no-till system to be 36.8 Mg C ha^{-1}; greater than that from a conventional tillage system at 29.8 Mg C ha^{-1} (Alvaro-Fuentes *et al.*, 2012).

Research has shown the SOC content in deeper layers of the soil is greater under conventional management practices (de Vita *et al.*, 2007). As a result of this shallow soil carbon storage in reduced- and no- tillage systems, if conventional tillage practices are re-employed in the short-term, e.g. during crop cultivation, the sequestration benefits are largely lost, resulting in little or no benefit over a complete rotation cycle (Smith *et al.*, 1998; Blanco-Canqui *et al.*, 2012). Furthermore, the mixing of cover crop residues into the soil causes organic matter to mineralize, with CO_2 emissions peaking just after tillage (Alvaro-Fuentes *et al.*, 2008a) and, hence, these soils may in the short-term become a source of CO_2 (Dersch and Böhm, 2001). In contrast, the annual incorporation of cereal residues into the soil was identified by Triberti *et al.* (2008) as the most effective method to sequester a significant amount of CO_2 in soils in Europe. Finally, soils under conservation agriculture only sequester carbon for a finite period, i.e. until soil carbon content reaches a new equilibrium (Smith *et al.*, 1998). The timeframe for this has been estimated to be 20 years after conversion from traditional practices (West and Post, 2002).

Research has shown that the carbon sequestration benefit for no-till practices using cover crops is greater than that for those without, able to store between an additional 0.10-1.0 Mg ha^{-1} y^{-1} SOC compared to a system without cover crops (Blanco-Canqui *et al.*, 2012). Similarly, Lehuger *et al.* (2011) studied the impact of leaving crop residues on the soil in monitored cropping systems in France and Germany, finding that although this increased soil respiration, the return of organic residues to the soil increased its organic carbon content by 265 kg C ha^{-1} y^{-1}. The carbon storage benefits from conservation agriculture can be increased by using legumes or perennial grasses in a no-till row crop rotation (Blanco-Canqui *et al.*, 2012). The deep-root systems of these crops cause carbon to be stored deeper in the soil profile, hence, reducing the stratification of soil organic carbon (discussed above), and improving long-term soil carbon sequestration (Blanco-Canqui *et al.*, 2012). It is, however, important to note that the carbon sequestration potential of cover crops is influenced by the species of cover crop, soil type, and weather conditions among other factors (Desjardins *et al.*, 2005; Blanco-Canqui *et al.*, 2012).

In the short-term, no-tillage soils have a low soil CO_2 efflux, and this practice is a better way of managing soil C than conventional agriculture techniques (Alvaro-Fuentes *et al.*, 2008a). In addition, significant increases in SOC after the adoption of conservation management are found to be relatively rapid, with Munoz *et al.* (2007) reporting a period of only two years for this, and the differences between conventional and no-till practices increasing in the long-term (Munoz *et al.*, 2007). It may also be interesting to note that the amount of fertiliser applied to a no-tillage plot can affect its carbon flux. For example, it was found that for a no-tillage plot 335 kg N ha^{-1} y^{-1} fertiliser was the optimum amount to achieve the greatest CO_2 mitigation (West and Marland, 2003). In contrast, when a substantially lower amount of fertiliser was applied to the field, the system became a net contributor to the atmospheric CO_2 pool, but was still associated with reduced emissions compared to conventional management practices (West and Marland, 2003). Similarly, a modelling study of the Spanish Mediterranean has concluded that the adoption of no-tillage practices in conjunction with high levels of N fertilisation (60-120 kg N ha^{-1}) has the potential for significant carbon sequestration in agricultural soils and, hence, offsets a proportion of the CO_2 emissions (Alvaro-Fuentes *et al.*, 2012). It is possible that adopting a practice such as no-till may reduce the amount of carbon released via soil erosion under conventional management practices and, therefore, Desjardins *et al.* (2005) note that no-till management will sequester soil carbon if this either reduces the rate of decomposition of soil carbon (Blanco-Canqui & Lal, 2008) or increases crop yield (which may arise from increased soil moisture) and, hence, carbon inputs into the soil.

One study from the review suggests that, depending on a number of factors, carbon sequestration in some no-till systems has the potential to be overestimated or may even be negligible (Constantin *et al.*, 2010). It is therefore important to examine each site specifically before employing no-till as a mitigation option. This complication is highlighted by King *et al.* (2004), who examined the effect of no-tillage practices in the UK, finding that when taking only carbon into consideration, no-till resulted in three times less sequestration compared to reduced tillage. However, when other GHGs were included in the analysis, the authors calculated a three-fold increase in carbon saving potential from reduced- as opposed to no- tillage. This change is due to a potential increase in N_2O emissions in no-tillage systems, which is not expected to occur in minimal tillage. It is then important to note that in contrast to the conclusions of a number of studies, the adoption of no-tillage as a mitigation strategy may in some cases increase GHG emissions (King *et al.*, 2004).

The highest emissions of CO_2 from the soil have been related to high soil moisture content, hence, it has been shown that climatic conditions impact on soil CO_2 emissions, with moisture and temperature being particularly important variables, altering CO_2 through its effects on vegetation growth and the activity of micro-organisms (Carbonell-Bojollo *et al.*, 2011). Although minimal soil disturbance associated with conservation agriculture practices has been shown to increase carbon storage, studies show that this may also result in higher N_2O losses (Ball *et al.*, 1999), the accumulation of mineral N in the soil profile, and associated environmental problems (Alvaro-Fuentes *et al.*, 2012). Research has shown that the majority of the net global warming potential of no-tillage systems stems from the N_2O emissions and, hence, it is important to improve N management in farm systems in order to benefit from the increased C storage that this management practice provides (Six *et al.*, 2004).

On a European scale, it has been estimated that the widespread conversion to no-till agriculture practices would sequester approximately 23 Tg C y^{-1} in the European Union, or 43 Tg C y^{-1} across wider Europe (Smith *et al.*, 1998). A number of studies have also been conducted on a country-wide scale, for example, Dersch and Böhm (2001) calculated that in Austria the conversion to no-tillage practises on steppe soils (i.e. phaeozems, chernozems, and kastanozems) could store the equivalent of 0.6% of the country's present annual CO_2 emissions. For Spain, a study of a number of long-term experiments estimated that the adoption of no-tillage practices would sequester around 0.14 Tg Cy^{-1} comparable to 1.1% of all agricultural CO_2 generated in the country during 2006 (Alvaro-Fuentes and Cantero-Martinez, 2010). Similarly, the adoption of reduced tillage practices would also sequester carbon, but to a lesser extent, with 0.08 Tg C sequestered per year, or 0.6% of Spanish agricultural emissions in 2006 (Alvaro-Fuentes and Cantero-Martinez, 2010). It therefore appears that in terms of carbon sequestration, no tillage practices offer the best mitigation potential. The scale of practice change, i.e. the extent of reduction in soil management, for example, from conventional tillage to no-till or to reduced tillage also has an impact on soil carbon. A study of agricultural plots in Denmark found that compared to conventional tillage, reduced tillage lowered net GHG emissions by an average of 0.56 Mg CO_2 eq ha^{-1} y^{-1}, whereas direct drilling resulted in a substantially greater emissions saving of 1.84 Mg CO_2 eq ha^{-1} y^{-1} (Chatskikh *et al.*, 2008).

It appears that the mitigation potential for conservation agriculture practises such as no-till is rather complex, being strongly influenced by climate and soil type in addition to a number of other factors (Desjardins *et al.*, 2005).

Carbon sequestration in soils can also be undertaken by promoting the input of organic materials on arable land rather than grassland, the introduction of perennials on arable set-aside land, by promoting organic farming, raising water tables (as there is less carbon loss from peats), and with restriction cropland management such as zero tillage (Freibauer *et al*., 2004). Areas need to be selected where there is high carbon sequestration potential as due to regional soil variations, intensification of farming and high uncertainties, it is very difficult to determine whether farming would still make a profit (Freibauer *et al*., 2004, Gaiser *et al*., 2009). Policy measures (e.g. CAP measures, set-aside land, subsidies) and land management policies that take into account climate change may aid more effective carbon sequestration measures within the agricultural sector (Freibauer *et al*., 2004).

Crop type

Mitigation can occur through crop and resource management, as crop genetic enhancement and the type of crop cultivated will affect CO_2 and N uptake as well as leaching. In a Norwegian farming study, Bonesmo *et al.* (2012) calculated the following GHG intensities for crops: barley (2442 kg CO_2 eq ha^{-1}), oats (2483 kg CO_2 eq ha^{-1}), spring wheat (2960 kg CO_2 eq ha^{-1}), winter wheat (3505 kg CO_2 eq ha^{-1}) and oilseed (2551 kg CO_2 eq ha^{-1}), although the values varied according to the soil, farming practices, etc. Nitrogen uptake and leaching may also be affected by the timing of sowing, with a delay in sowing reducing N uptake during autumn and winter, increasing leaching (Olesen *et al.*, 2004). This will not only affect mitigation, but also lead to higher costs through increased fertiliser usage.

Crop management practices

Reducing nitrous oxide (N_2O) emissions is a key mitigation action and Ortiz *et al.* (2008) suggest that these can be halved in intensive irrigated systems without affecting wheat yields, provided the correct amount and timing of nitrogen applications are given. Similarly, Górny *et al.* (2011) highlight an increasing importance of breeding winter wheat (*Triticum aestivum L.*) cultivars with better adaptation to lower, more optimised nitrogen fertilization regimes. They found that while it was possible to identify cultivars which were both efficient under low and enhanced nitrogen regimes, not all could cope with both situations.

Water management

It has been suggested that changing water management practices has the potential to alter the effectiveness of agricultural mitigation and adaptation options (Falloon and Betts, 2010). It is thought that on balance under climate change, irrigation leads to an increase in soil organic carbon, with potential increases in productivity thus contributing to mitigation, although the interactions are poorly understood (Falloon *et al.*, 2009). Water intensification, if poorly designed, located and managed could lead to increased soil erosion, with concomitant organic carbon and nutrient losses (Falloon *et al.*, 2009).

4.1.2 Agriculture in China

So far, research on agricultural methods of mitigation of climate change overwhelms agriculture adaptation in China. On the basis of the reviews (Zou *et al.*, 2011; Li *et al.*, 2012; Zhang *et al.*, 2012), existing mitigation practices in agriculture in China can be classified into three categories (Table 5); mitigation practices aiming to reduce GHG emission directly; soil carbon storage practices aiming to boost soil carbon sequestration; and storage and integrated practices aiming to attain the objectives of GHG mitigation and carbon storage simultaneously.

Table 5: Classification of existing mitigation practices in agriculture in China.

Type	Practices	Is also employed as an adaptation measure?
Mitigation practices	Nitrification inhibitor	Not yet
	Methane inhibitor	Not yet
	Slow-release fertilizer	Possible
	Manure storage	No
Soil carbon storage practices	Conservation/no tillage	Possible
	Crop residue management	Yes
	Manure management	No
Integrated practices	Water management	Yes
	Nitrogen fertilizer management	No
	Land management	Yes
	Cover crop	No
	Varieties of crop planting/animal	Yes
	Feed management	No

Sources: Zou *et al.* (2011); Li *et al.* (2012); Shi *et al.* (2012); Zhang *et al.* (2012).

4.2 Biodiversity

No specific mitigation measures were found for biodiversity, but mitigation in many of the other sectors involves various aspects of managing biodiversity and soils, to enhance storage or prevent losses. This may be direct as in the case of afforestation or indirect as with the re-creation of wetlands to manage flooding, which have the additional benefit of enhancing carbon storage (see Section 4.3 below).

4.3 Coasts

As far as mitigation is concerned, the majority of studies were theoretical with regards to considering the amount of carbon sequestered in coastal wetlands or saltmarsh systems. Numerous studies in this review emphasized the importance of coastal wetlands as a global carbon sink, reducing levels of atmospheric GHGs (Burkett and Kusler, 2000; Choi *et al.*, 2001; Connor and Chmura, 2001; Jickells *et al.*, 2003; Irving *et al.*, 2011; Kirwan and Blum, 2011). For example, it was found that the restoration of coastal marsh could be a more efficient method per unit area at removing carbon from the atmosphere than afforestation (Choi *et al.*, 2001; Trulio *et al.*, 2007). With this knowledge, wetland restoration and conservation for climate mitigation appears to be highly recommendable, reducing global levels of anthropogenic GHG emissions (Moseman-Valiterra *et al.*, 2011). This approach is particularly viable, as globally coastal vegetation removes and stores carbon from the atmosphere at a rate of 60-210 t C km^2 y^{-1} which is far greater than that of any similar terrestrial system, with evidence to suggest that created marsh sequesters higher amounts of carbon than natural wetland (Hansen, 2009; Irving *et al.*, 2011; Kirwan amd Blum, 2011). In the face of climate change, coastal marsh continues to be a favourable option, with sea level rise leading to increases in areal marsh coverage and elevation; warmer temperatures increasing plant growth rates; and increases in the carbon storage capacity of marsh soils (Connor *et al.*, 2001; Choi and Wang, 2004; Irving *et al.*, 2011). Andrews *et al.* (2008) calculated that land reclamation for pasture along the Humber Estuary since the 1700s has prevented the storage of 320,000 tonnes of organic carbon, whereas the restoration of tidal flow as part of managed realignment schemes in the Humber and Blackwater estuaries could sequester a total of 38.4-3597.1 t C y^{-1} and 21.7-639.49 t C y^{-1} respectively, depending on future policy scenario (Luisetti *et al.*, 2011). Coastal wetland creation and restoration provide long-term mitigation benefits impacting on a global scale, with carbon in saltmarsh being stored in below-ground biomass for decades (Hansen, 2009; Irving *et al.*, 2011).

4.4 Forests

The main characteristic mitigation option for forests is carbon sequestration and storage in trees, vegetation and soils (Garforth, 2012). According to Patosaari (2007), forests contain about 1.2 trillion tonnes of carbon, which is much higher in comparison with the corresponding stored quantities in all terrestrial vegetation and soils. Moreover, forests and wetlands exhibit the highest capacity for the provision of long-term sequestration of carbon (MEA, 2005). A typical carbon density ranges from 40 to 60 Mg C ha^{-1} for boreal forests, 60 to 130 Mg C ha^{-1} for temperate forests, 120 to 194 Mg C ha^{-1} for tropical forests, and about 250 Mg C ha^{-1} for rainforests. Dixon *et al.* (1994) estimated that approximately two-thirds of the terrestrial carbon in forest ecosystems is contained in soils. Moreover according to Lal (2005), the soil carbon stock may comprise as much as 85% of the terrestrial carbon stock in the boreal forest, 60% in temperate forests and 50% in rainforests. Mangroves also have significant storage of carbon (Alongi, 2002) and the capability to absorb approximately 25.5 millions tonnes of carbon per year (Ong, 1993). Furthermore, there is a distinction between growing and well-managed forests which are considered an effective way of carbon storage in comparison with old-growth forests, which may emit carbon due to the onset of decomposition (Patosaari, 2007). Indicatively, Georgia's forests absorb a volume of CO_2 equal to 25% of the country's gross CO_2 equivalent GHG emissions in 2000, while for Azerbaijan the corresponding percentage was about 7%-8% (MNP-AM, 2010; MENR-AZ, 2010; MEPNR-GE, 2009).

The contribution of carbon sequestration to climate change mitigation can be confirmed within the country data reported in Second National Communications reports submitted to the UNFCCC and Table 6 provides an overview of the implemented projects within the framework of the CDM mechanism, which are related to the forest sector and contribute to the mitigation of the climate change through significant CO_2 reductions on an annual basis. This shows that many European countries are participating in the CDM mechanism as a means of mitigation, although in Europe only Albania is a host party.

Table 6: A selection of implemented mitigation projects within the framework of the CDM mechanism of the forest sector (Source: UNFCCC).

Registered	Title	Host Parties	Other Parties	Tonnes CO_2 eq reduction per year
02-Jan-10	Assisted Natural Regeneration of Degraded Lands in Albania	Albania	Canada, Italy, Luxembourg, France, Japan, Spain	22,964
15-Jan-10	The International Small Group and Tree Planting Program (TIST), Tamil Nadu, India	India	UK	3,594
21-Jul-10	Reforestation as Renewable Source of Wood Supplies for Industrial Use in Brazil	Brazil	Netherlands, Italy, Luxembourg, France, Ireland, Switzerland, Japan, Spain	75,783
15-Sep-10	Reforestation on Degraded Lands in Northwest Guangxi	China	Switzerland, Ireland, Spain	87,308
07-Jan-11	AES Tietê Afforestation/Reforestation Project in the State of São Paulo, Brazil	Brazil	Canada, Italy, Luxembourg, France, Japan, Spain	157,635
11-Feb-11	Reforestation of grazing Lands in Santo Domingo, Argentina	Argentina	Switzerland	66,038
04-Apr-11	Kachung Forest Project: Afforestation on Degraded Lands	Uganda	Sweden	24,702
07-May-11	Southern Nicaragua CDM Reforestation Project	Nicaragua	Canada, Italy, Luxembourg, France, Japan, Spain	7,915
26-May-11	Forestry Project in Strategic Ecological Areas of the Colombian Caribbean Savannas	Colombia	Spain	66,652

Nevertheless according to Seppala *et al.* (2009), it is vital to support scientific research regarding management and policy measures, because they contribute to the enhancement of adaptation and mitigation practices. Such research can include the identification of specific timber species that are both more resilient to climate change and can store significant quantities of carbon. Moreover, the local influences of trees and forests on the hydrological cycle must be examined thoroughly to flag all these cross-sectoral issues (FAO, 2012), while special efforts must be given for the planning of effective carbon sequestration measures taking into consideration the negative effects on biodiversity, genetic resources and water. Moreover, further scientific research is needed in order to:

1. Evaluate the cost-effectiveness of adaptation and mitigation measures;
2. Adopt more intensive sylviculture treatments on biomass production;
3. Assess their potential for mitigation;
4. Examine the potential effects of more intensive techniques which can lead to increased levels of carbon sequestration;
5. Evaluate the substitution effect of wood products;
6. Promote efficient measures for stimulation of policies to support sustainable use of wood;
7. Provide incentives for multiple uses of wood and wood products; and
8. Introduce the proper socio-economic frameworks in order to lead to innovative forest management practices focusing on the role of policy-making and the contribution of forest owners (Standing Forestry Committee Ad Hoc Working Group III on Climate Change and Forestry, 2010).

4.5 Urban

Section 3.5 highlighted that many of the urban measures can be considered as having both adaptation and mitigation benefits. The most notable being urban street trees and those primarily implemented for mitigation purposes will now be discussed.

4.5.1 Urban trees

Urban trees sequester carbon directly, both in the trees themselves and in the soils and, hence, can be seen as mitigating climate change by reducing atmospheric CO_2 concentrations. Despite this, urban trees sequester carbon at less than half the density of natural forest (Nowak and Crane, 2002) and, hence, in mitigation terms, the amount of carbon sequestered by urban trees is therefore seen as negligible; not of great enough magnitude to achieve local GHG reduction targets (Pataki *et al.*, 2011).

Research into the amount of carbon storage provided by urban trees in Leicester found that the amount of carbon stored in aboveground biomass in the city totals an estimated 231,521 t, equal to a density of 3.16 kg C m^{-2} over the urban area (Davis *et al.*, 2011). Gardens in residential areas were found to store relatively little carbon, with the highest carbon density measured in areas of tree cover on public sites (Davis *et al.*, 2011). There remains potential to increase tree cover in the city, with it estimated that if 10% of the present grassland owned by the City Council were planted with trees, an extra 28,402 t C would be sequestered into the current pool (Davis *et al.*, 2011).

In a study of urban forests, it was found that those consisting of natural pine-oak forests, mangroves, and stands of highly invasive trees achieved the greatest levels of CO_2 storage (Escobedo *et al.*, 2010). The direct carbon sequestration by urban trees in these cities is able to offset 2.6% and 1.6% of city-wide CO_2 emissions respectively (Escobedo *et al.*, 2010).

These results suggest the effectiveness of urban trees to offset emissions in these cities is moderate, with the relative reductions in emissions comparable to those achieved by existing policies to reduce CO_2 emissions (Escobedo *et al.*, 2010). Despite this performance, it was found that further increasing the tree area would not make a substantial contribution to emissions reductions (Escobedo *et al.*, 2010).

Street trees can be associated with a number of adverse effects, planting and maintenance costs (McPherson and Rowntree, 1993; Tallis *et al.*, 2011). Planted species vary in their suitability to function in the urban area, for example, tree species producing fruit, flowers or seeds, such as the maple function poorly, littering surfaces, being potentially hazardous to both pedestrians and vehicles (Merse et al., 2009; Hegedüs *et al.*, 2011). Furthermore, there often exists concern over potential damage to building foundations when street trees neighbour properties (Rotherham, 2010).

Conditions in the urban environment for tree survival are harsh, with space and moisture restrictions for roots being problematic, and many street trees being in poor condition (Schröder, 2008). To increase tree survival in the German city of Osnabrück, measures are being taken to improve site conditions for growth (Schröder, 2008). These include the construction of root ducts and chambers underneath traffic lanes to increase the rooting zone for trees to around 15 m^3 per specimen (Schröder, 2008).

4.5.2 Green roofs

In terms of mitigation, green roofs can make a small contribution to carbon sequestration, with CO_2 stored both in plant tissues and the soil substrate (Rowe, 2011). The magnitude of carbon storage achieved by twelve 2.84 m x 4.6 m green roofs in Michigan was calculated at 375 g C m^{-2}, with the average roof storing approximately 162 g C m^{-2} in the aboveground biomass alone (Getter et al., 2009). The effectiveness of green roofs to act as a sink can be improved by altering the species selection, depth of substrate and its composition, and by improvements in management (Rowe, 2011). It is also important to note that the potential for carbon sequestration is somewhat limited, as over time a green roof system will reach a carbon equilibrium, and no longer function as a sink for carbon (Rowe, 2011).

4.5.3 Urban intensification

Urban intensification, or densification, is seen as one possible way to mitigate climate change (Melia, 2011). Increasing density and confining expansion of the urban area both increases accessibility reducing the need to travel, and concentrates the demand for public transport (Bunce, 2004; Ancell and Thompson, 2008; Dodson, 2010; Melia, 2011). Reduced emissions from travel have global benefits, mitigating atmospheric GHG concentrations (Williams, 1999; Melia, 2011).

A modelling study for the city of Copenhagen found that the geometric design of cities can have a large impact on energy use (Strømann-Andersen and Sattrup, 2011). The geometry of urban canyons was able to alter energy consumption in offices by as much as 30%; and by 19% in residential dwellings (Strømann-Andersen and Sattrup, 2011). The study found that a higher urban density decreases demand for cooling over the summer as a result of shading, and correspondingly increases demand for heating during the winter season as solar gains are reduced (Strømann-Andersen and Sattrup, 2011). With a very effective low-energy design, the study found it is possible for an office building in a dense urban area to consume a minimal 70 kWh m^2 y^{-1} energy (Strømann-Andersen and Sattrup, 2011). Future urban designs may be able to utilise the reflective properties of façades to redistribute light in canyons in dense urban areas (Strømann-Andersen and Sattrup, 2011).

The city of Dublin was also found in this review to be adopting a policy of urban densification (Howley, 2009), and at a country-wide scale, the UK Government seeks to increase the number of new dwellings built on brownfield sites from 50 to 60%. Such re-developments are often more costly (see Section 15.1.5), but can be encouraged by tax incentive programmes for projects undertaken on brownfield sites. These aim to reduce the cost of such developments whilst improving the urban environment, forcing developers to clean up contaminated sites and can renew areas of the city centre in the process by preventing urban decay and inducing new economic activity (Bunce, 2004; Hayek *et al.*, 2010; Williams, 1999).

The increasing density of dwellings could also put existing amenities and services under a lot of pressure, overstretching the existing infrastructure which in older cities is already overcrowded (Dixon and Depuis, 2003; Williams, 1999) and, hence, new infrastructure may be required (Searle, 2010). The intensification of cities also runs the risk that the capacity of these areas could be broken (Williams, 1999).

4.5.4 Building materials

The use of building materials with minimal embodied energy can be considered as a form of mitigation, and is able to reduce both heating and cooling loads. The UK's Beddington Zero Energy Project provides a good example of this, with many of the construction materials being either recycled (e.g. reclaimed structural steel, recycled sand and aggregate) or sourced locally within 30 miles of the site (Chance, 2009). In addition, the dense concrete blocks used in construction have a high thermal mass, designed to maintain building warmth during winter months and cool during the summer (Chance, 2009).

Similarly, the design of the Earthship Brighton project also used a range of low-impact building materials, including eco-cement (one-third conventional cement, two-thirds reactive magnesia), reclaimed vehicle tyres which construct the earth-rammed tyre walls and act as a storage heater; accumulating heat over the day and releasing it at night (Ip and Miller, 2009).

4.5.5 Insulation

Improvements in insulation for walls, floor and roofs for the purpose of draught-proofing, and to reduce the loss or gain of excess heat, is one of the most effective measures for increasing energy efficiency (Xing *et al.*, 2011; Mavrogianni *et al.*, 2012). Examples include the installation of triple glazing to significantly reduce heat loss through windows and doors (Zimmerman and Anderson, 1998; Holmes and Hacker, 2007; Chance, 2009). A low energy residential estate in Warsaw is, in addition to the above, further insulated by aluminium blinds on windows and balcony doors which significantly reduce heat loss (Wojdyga, 2009).

Preliminary modelling results show that roof insulation has the potential to reduce heating load in Cyprus by over 45% and 75% in summer and winter, respectively (Florides *et al.*, 2000). In the building simulation with roof insulation, room temperature did not exceed 40°C in the summer, whereas in the case without, indoor temperature rose to 46°C (Florides *et al.*, 2000). Similarly for winter, simulated temperature in the room with an insulated roof was 4°C warmer than without, being closer to the required ambient room temperature (Florides *et al.*, 2000). A modelling study for a building in the city of London found that roof insulation and fenestration improvements were able to reduce daytime temperature in the living room area over the warmest 5-day period of modelling by an average of 0.76°C, with a 1.30°C maximum (Mavrogianni *et al.*, 2012). However, insulation does not lead in all cases to reductions in summer ambient room temperature. For example, the insulation of walls and floors were modelled to increase indoor daytime temperatures over the hottest 5-day consecutive period, with a combined increase in temperature of 0.46°C and a maximum of 0.71°C. Hence, in this case, if passive measures such as night-time ventilation are not employed, insulation improvements could in fact increase the risk of overheating (Mavrogianni *et al.*, 2012).

Future directions to improve building energy efficiency with insulation concern the development of new insulation materials such as aerogel, multi-layer insulation and transparent insulation materials, which have a solar energy transmittance of over 50% and a low thermal conductivity (Xing *et al.*, 2011).

4.5.6 Example of a low-energy retrofit

In Sandwell, UK, a recent retrofit of a new museum building has been completed to convert it into a low energy building titled THEpUBLIC (Battle *et al.*, 2006). This is a flagship building which is part of the European MUSEUMS project; optimising energy efficiency and sustainability in nine museum buildings across Europe and aiming to increase acceptance of renewable technologies and sustainable architecture in public spaces (Battle *et al.*, 2006). Targets for the building were a 40% reduction in energy consumption and maintenance along with a significant reduction in embodied energy from construction materials (Battle *et al.*, 2006). This development incorporates building measures such as daylighting, a mixed-mode ventilation system, and an intelligent façade system with external shading and natural ventilation (Battle *et al.*, 2006). THEpUBLIC development emphasises increased water efficiency, with devices to limit flow and reduce potable water demands, rainwater harvesting, greywater re-use, efficient taps and special flushing systems. Furthermore, there exists the potential to expand the development through the use of renewable energy collection systems, such as solar thermal collectors for hot-water heating (Battle *et al.*, 2006). These measures enable the building to achieve total energy savings greater than 35%.

4.5.7 Sustainable transport systems

Sustainable transport is an aspect highlighted in many low-energy developments. For example, at the Beddington Zero Energy Development (BedZED), and the Nydalen development in Norway, living streets make the areas more suitable for pedestrians and cyclists (Chance, 2009; Høyer, 2009). BedZED also has strong public transport links, with a bus-stop and two train stations nearby, and the low number of car parking spaces encourages the use of sustainable forms of transport (Chance, 2009).

Car-sharing services are also included in a number of low-energy community developments, including that at Nydalen (Høyer, 2009), and again BedZED (BioRegional, 2002). The community car-club at BedZED allows residents to pay per mile for use of cars owned by a company; hence, they do not require their own car for transport (Chance, 2009). As a result of the above measures, BedZED communities have seen a significant reduction in car use at 54% below the local average, and a high bike ownership of over 50% (Chance, 2009).

4.5.8 Green energy

In this review numerous examples were found of solar energy being used in housing developments of various scales and design throughout Europe. This energy can be used for various applications, for example solar electricity, also termed solar photovoltaic (PV) power; and solar thermal systems which collect heat energy from the sun for household use such as hot water heating.

Both applications have been used extensively at a housing development at Nieuwland in the Netherlands. Here, PV panels were installed on the roofs of 900 houses and solar collectors enable the use of solar radiation as energy to power the hot water and central heating systems (Anon, 1996). The use of solar energy in this development enables home-owners to save an average of 275 m^3 gas per year; and the homes in Nieuwland are able to generate around 1,800 kWh of renewable energy annually (Anon, 1996). In terms of the economics of solar PV systems, the rents for homes with solar energy do not exceed those of standard homes; however they do offer a financial incentive, with savings in heating requirements equivalent to 200-275 m^3 gas per year (Anon, 1996).

Another development maximising the use of solar energy is Borgo Solare, a residential settlement in Italy (Aste *et al.*, 2010). The installation of solar thermal and PV systems is now mandatory in this urban district, with PV systems in single-family apartments with a power rating of 2 kWp, and 4 kWp for multi-family apartments (Aste *et al.*, 2010). Solar thermal systems have been very effective here, able to supply over 50% of annual household hot water requirements in the district (Aste *et al.*, 2010). Solar collectors have also been used in a polish low-energy residential development (Wojdyga, 2009), and in the flagship zero energy project at Beddington, UK (BedZED, Chance, 2009). In addition to this, at BedZED solar PV cells cover 777 m^2 of the roof area to supply approximately 20% of the total electricity demand of the complex (Chance, 2009). The solar panels and low-energy building here have reduced CO_2 emissions by over half with respect to the average UK home (Chance, 2009).

Other examples of solar PV cells on building façades and roofs include the Stadtweke Halle in Westfalen; the Wernberg plant of Flachglas AG., Germany where PV cells covers an area of 140 m^2 of the façade; and a 500 m^2 PV façade area of the ELSE building in Ispra, Italy (Benemann and Chebab, 1996). Solar energy systems also have the potential to be installed on a hostel roof in Milan, with a combination of solar PV and solar thermal systems simulated to meet a substantial proportion of electricity demand for lighting and appliances (Adhikari *et al.*, 2011). The replacement of a traditional gas condensing boiler with two solar thermal collectors for hot water heating on the roof garden could save over 0.5 t CO_2 per year (Adhikari *et al.*, 2011). In addition, a solar PV system of 22 modules could be installed on the rest of the available roof space to produce around 5.3 MWh electricity per year resulting in annual CO_2 savings of 2.3 tonnes (Adhikari *et al.*, 2011).

A number of studies have assessed the feasibility of renewable energy in the UK. These include an assessment for the Eastside area of Birmingham commissioned by the UK Carbon Trust under the direction of Birmingham City Council (Jefferson *et al.*, 2006), and an investigation into the potential of various micro-generators for UK households (Allen and Hammond, 2010). Both studies found wind turbines (micro-wind turbines for households) to be unsuitable, either as a result of low wind-speeds, sensitivity to site location, or problems with planning regulations (Jefferson *et al.*, 2006; Allen and Hammond, 2010). Solar PV cells were identified as being the most suitable, able to reduce CO_2 emissions by over 35% at Eastside, producing an estimated 135 kWh m^2 y^{-1} (Jefferson *et al.*, 2006) and 27-57% of electricity demand in UK households (Allen and Hammond, 2010). PV systems in these studies were found to result in the greatest emissions reductions, with savings of 830-1,300 kg CO_2 eq y^{-1}, compared to the micro-wind turbine which offset an average of 79-122 kg CO_2 eq y^{-1} only (Allen and Hammond, 2010). A number of small-scale low energy developments in the UK, including the Leigh Park site in Hampshire (Bahaj and James, 2007) and a small development of 14 homes at Lingwood, Norfolk (Monahan and Powell, 2011) utilise solar energy. Solar PV has made a significant contribution to annual electricity demand at both sites (Bahaj and James, 2007), supplying between 17 and 27% of total annual energy demand for homes at Lingwood; meeting 65-89% of hot water energy demand, and 14-41% of electricity consumption (Monahan and Powell, 2011). These buildings have been certified as Level 3 according to the Code for Sustainable Homes and have energy consumption substantially less than the average UK Household, with for example 4% less electricity and 66% less gas consumption (Monahan and Powell, 2011). Solar technology in this development reduces the running costs of homes to around 35% below the regional average (Manahan and Powell, 2011). Homes at Lingwood also have on average 47% lower CO_2 emissions, and an offset for grid generated electricity of 60% (by solar PV), and 40% of gas (by solar thermal) (Monahan and Powell, 2011).

4.5.9 Low energy residential settlement case studies

One of the flagship zero-energy projects in the UK, already discussed above, is the Beddington Zero Energy Development (BedZED) in south London. This is a high density residential development which forms the UKs largest mixed-use sustainable community, consisting of 220 residents and 100 office workers (BioRegional, 2002; Chance, 2009). The project is located on a brownfield site on the edge of the city, and takes a holistic approach; reducing embodied emissions during construction, greener transport links, reducing waste, creation of allotments for local food production, increasing quality of life of residents, and improving the local economy (Chance, 2009). Building use in the project is mixed, with residential dwellings, office-space and a mix of income groups (Chance, 2009). The average resident of the BedZED has an ecological footprint of 4.67 global hectares, compared to the 6.3 ha average for the UK (BioRegional, 2002; Chance, 2009). BedZED has a number of low energy building design features and materials, with passive ventilation, daylighting, energy efficient appliances, and smart-meters (Chance, 2009). As a result of these installations, residents see over an 80% reduction in energy use for hot water, and 45% for electricity (Chance, 2009). The target was to reduce heating, cooling and ventilation energy demands by 90% compared to the average UK residential dwelling (Chance, 2009). The BedZED community is increasingly self-sufficient, with allotments producing food sold at the local produce market, and a vegetable box scheme (Chance, 2009). Actors in this development include the Peabody trust, the BioRegional Development Group and the ZEDfactory, and the scheme has support from the local government helping to achieve the UK's obligations for a reduction in GHG (Chance, 2009). If implemented on a European-wide scale, it is estimated that reductions of 90% of CO_2 emissions could be achieved without adverse impacts on quality of life of residents (Chance, 2009).

A sustainable district has also been built in the Norwegian municipality of Nydalen, Oslo, on a 500 acre brownfield site (Høyer, 2009). The derelict industrial buildings at the site were regenerated with retrofits, and a number of new dwellings were also constructed on-site, as well as the newly constructed office building, "Pynten" (Høyer, 2009). This building has a low-energy design resulting in energy use (80 kWh m^{-2}) and heating requirements as low as 25 kWh m^{-2} (Høyer, 2009). The project as a whole comprises mixed use, with 200 small industrial firms on site as well as housing, designed to meet the requirements of residents with local amenities and services reducing the need for travel (Høyer, 2009). All buildings in the development have a high energy efficiency rating, with average heat energy requirements of around 160 kWh m^{-2} (Høyer, 2009). All new constructions on the site receive energy via a central heat pump, which is the largest of its type in Europe; with heat transferred away from the surface into rock formations during the summer to cool the buildings (with a 7.5 MW potential cooling capacity), and transferred upwards during the winter to provide heat (9.5 MW heating capacity) (Høyer, 2009).

In Italy, a new low-energy residential settlement has been built in the urban district of Borgo Solare by the private company Gambala Immoviliare, with research and development support from the Politecnico di Milano (Aste *et al.*, 2010). All dwellings on site are designed to be energy saving; with high energy efficiency, with the use of renewables (Aste *et al.*, 2010). The environmentally conscious design of this development extends to the choice of building materials, which have a low environmental impact; taking embodied energy into account, without incurring substantial additional final cost (Aste *et al.*, 2010). This scheme complies with the European Directive on Energy Performance of Buildings (EPBD), using a combination of natural ventilation, daylighting, heating ventilation and cooling systems, solar thermal collectors, solar PV, and ground source heat pumps (Aste *et al.*, 2010). All buildings in the Borgo Solare development comply with European Standard for the production of domestic hot water, and in theory over 50% of a dwellings annual hot water requirements are met by the solar panels (Aste *et al.*, 2010).

These studies highlight the success of existing low-energy developments, and the potential for similar schemes to be implemented across Europe for the purposes of climate change adaptation and mitigation.

4.6 Water

Mitigation to help alleviate climate change relates to the supply of water resources, including (1) the creation of wetlands, allowing for (2) carbon sequestration and subsequent carbon storage, as well as (3) mitigating the climate. The store of carbon is dependent on the rate of decomposition, leaching and erosion (Ostle *et al.*, 2009). Since the carbon sequestration ability of soils is moisture dependent (Lamparter *et al.*, 2009), there can be a high absorption rate achieved by coastal and riverine wetlands. The sequestration of carbon depends on whether existing stocks and the ecosystems that sustain them can be maintained, and whether additional soil carbon can be added. Changing land use can result in a rapid loss of carbon from peatlands, grasslands, plantation forest and native woodland. The need for land for farming and renewable forms of energy could have significant impacts on the carbon store in the UK (Ostle *et al.*, 2009) and elsewhere. Mitigation could also mean the protection of land, such as peatland and other organic soil carbon stocks. Hence, how we use the land is very important and can influence the balance between new and existing carbon storage (Ostle *et al.*, 2009). Future land use management is important, particularly due to the uncertainty associated with carbon storage.

Many of the mitigation measures associated with water resources depend on the land, so are highly influential in the agricultural and forestry sectors, and to an extent the coastal sector (due to wetlands). Interlinkages between the sectors are complex, and whilst there are some excellent examples of present ideas and mitigation practices (e.g. Falloon and Betts, 2010), many of the studies relate to future projections. As time progresses and more carbon mitigation schemes occur, further evaluation of the efficiency of these schemes is required.

5. Cross-sectoral interactions within and between mitigation and adaptation

This review has revealed a lack of clarity in how adaptation and mitigation synergies and conflicts (antagonisms) have been viewed in papers, grey literature and in policy documents. The main confusion concerns whether the relationship of interest is between adaptation and mitigation measures and climate change impacts, or between the measures themselves. Here we propose a set of definitions to overcome this confusion.

Cross-sectoral interactions are the ***impacts*** that adaptation and mitigation measures in a given sector have on another sector. These interactions can have a number of different outcomes and may be neutral, (primarily) positive, (primarily) negative or mixed (Berry *et al.*, 2008b; Berry and Paterson, 2010). It should be highlighted here that positive interactions contain (1) simple positive interactions (i.e. those which do not directly affect adaptation or mitigation efforts in the given or another sector) and (2) synergistic interactions (e.g. those which enhance the ability of the given or another sector to adapt to, or mitigate, climate change). The same is true for negative interactions, which consist of (1) simple negative interactions, and (2) antagonistic interactions. The neutral category is the smallest, as it is rare that a measure has no effect on other sectors although there are, of course, within sector impacts.

Adaptation interventions are commonly designed to impact on a particular sector. Some examples for coasts being the construction of seawalls and breakwaters at Ria de Aviero, Portugal, to reduce beach erosion (da Silva and Duck, 2001); coastal wetland restoration schemes in the UK to offset habitat loss from coastal squeeze (Dixon, *et al.*, 1998; MacLeod *et al.*, 1999; Pethick, 2002; WWF, 2002; Winn *et al.*, 2003); and the construction of dams, sluices and storm-surge barriers in the Netherlands to reduce vulnerability to future sea-level rise and storm-surge events (Elgershuizen, 1981; Saeijs and Stortelder, 1982; Wolff, 1992; Schekkerman *et al.*, 1994; Smits *et al.*, 2006).

As with the CLIMSAVE IAP, it is important to stress that the cross-sectoral nature of many adaptation measures means they will almost certainly impact on multiple sectors (Box 1). Despite this, few studies in this review went as far as to examine the wider implications of the adaptation measures, tending to focus on one sector, and failing to acknowledge many of the secondary effects. One example of this is the managed realignment policy in Essex. This review found numerous studies covering all aspects of the scheme, with Dixon *et al.* (1998) focusing on opportunities for habitat creation at the various sites; Garbutt *et al.* (2006) researching habitat development on the agricultural land to which tidal-influence was restored; French (2008) modelling the effectiveness of this approach as a flood defence; and MacLeod *et al.* (1999) quantifying geochemical changes at one site after implementation of the scheme. Not one of these studies covers all aspects of the realignment, however, by compiling the research it is possible to obtain a more detailed cross-sectoral overview.

Coastal changes also will have implications for water resources, as sea-level rise is expected to result in increased water tables and salinisation of groundwater and land (Kundzewicz *et al.*, 2007). Changes to tides and extreme water levels mean that saline waters could propagate further up rivers via the back water effect. In addition, climate change could affect the course of rivers and change local biodiversity (Delta Commissie, 2008; Box 1). This could have knock-on effects for drinking supplies, biodiversity and irrigation and therefore farming. Changes to inputs into the hydrological cycle can affect river discharge, and in turn this can affect sediment supply and availability downstream to the coast. Changes to the river – whether caused by climate change or adaptation measures (e.g. dam building, river management and the building of levees) – will affect the amount of sediment available and frequency of flooding (Hoque and Alam, 1997).

Similarly in the review of agriculture in China, although the listed adaptation interventions are generally intended to impact the agricultural sector, some practices will exert an influence on other sectors. Most obviously, irrigation practices will exert a marked influence on water consumption and quality, but other practices will also impact on multiple sectors. However, few of the studies reviewed examined the cross-sectoral implications of the adaptation measures outside of biodiversity and agriculture (Table 7), although there are some important interventions which will be discussed.

Table 7: Cross-sectoral interactions in agriculture in China.

Adaptation Intervention	Coasts	Biodiversity	Water	Urban	Forest	Agriculture
Water and irrigation infrastructure			×			×
Flood prevention infrastructure	×	×				×
Intra-basin water transfer projects		×	×	×	×	×
Water-saving irrigation			×			×
Varieties of crop planted		×				×
Planting time adjustment		×				×
Use of different species more suitable for climate changes		×				×
Conservation/no tillage		×	×			×
Weed and pest control		×				×
Terracing of sloping land						×
Water storage			×			×
Breeding selection		×				×
Genetically modified organics		×				×
Disaster early-warning system	×		×	×		×
Fertilizer management						×
Flood prevention standards	×		×			×

6. Neutral interactions

Most of the neutral interactions concerned adaptation in the urban sector to reduce temperatures, where strategies, such as white topping or building measures (e.g. Zimmerman and Anderson, 1998; Kleerekoper et al., 2012; Synnefa *et al.*, 2011), have no recorded direct effect on other sectors, although by reducing temperatures they may reduce the need for other adaptation and/or mitigation measures. It is estimated that if whiting is employed in all urban areas, considering the range of global cooling estimates, this could amount to a temperature reduction equivalent to 25-150 billion tonnes CO_2 (Akbari *et al.*, 2012).

There are very few other measures which come into the neutral category, although some biodiversity adaptation measures, many of which are site-based, such as assisted colonisation of species and management of protected areas (see Sections 3.2.1 and 3.2.6), have minimal impact outside the sector. This would not apply where extension to existing protected areas or new sites are proposed, as it would take land from other uses.

7. Positive interactions

This was the largest category of recorded cross-sectoral interactions; in terms of the number of sectors involved, as well as the impacts of adaptation and mitigation measures. Many of the interactions involved biodiversity or water. This section is divided into simple cross-sectoral interactions, i.e. where the impacts in the affected sector did not lead to any consequences for its adaptation and mitigation actions; and synergies, i.e. mitigation or adaptation strategies which themselves benefit mitigation and adaptation.

7.1 Simple positive cross-sectoral interactions

Simple positive cross-sectoral interactions identified in this review were found to concern water quality only. In the agricultural sector, for example, a study from Lebanon showed that early sowing of safflower increased yields and led to the capture of more residual soil N, reducing nitrate leaching into groundwater (Yau, 2007). For the coastal sector, evidence was found of examples where saltmarsh restoration led to improvements in local water quality (Chang *et al.*, 2001; Woodward and Wui, 2001; Darnell and Heilman, 2007), providing treatment of stormwater runoff, as well as being a sink for contaminants and nutrients (WWF, 2002; van Dyke and Wasson, 2005; Andrews *et al.*, 2008; Garbutt and Wolters, 2008). Shepherd *et al.* (2007) quantified the benefits of managed realignment as a nutrient sink, with realignment on the Blackwater Estuary resulting in the additional annual storage of 200-795 tonnes of nitrogen; and 146-584 tonnes of phosphorus. Another study found that wetland creation and the associated sediment accretion on the Humber would enhance the capacity of the estuary as a sink for contaminant metals, and subsequently help to improve the regional water quality of the North Sea (Jickells *et al.*, 2003).

Coastal adaptation options such as wetland restoration can also improve water clarity (Andrews *et al.*, 2006; Darnell and Heilman, 2007). Stormsurge barriers in the Netherlands, as well as decreasing tidal velocity in estuaries have been found to improve water clarity, which in turn could lead to an increase in primary productivity by phytoplankton (Elgershuizen, 1981). This would be an indirect benefit to biodiversity, and as such is not regarded here as a synergy.

Biodiversity strategies, such as the corridors being created in the Netherlands as part of the de Doorbraak project (Box 1; Section 8.2.1) have also led to improvements in water quality (WRD, 2011). Similarly for the forestry sector it was found that where planting occurs on former agricultural land, water quality (especially nitrate levels) and recharge may be restored to pre-agricultural levels (Plantinga and Wu, 2003).

7.2 Synergistic interactions

This review defines synergies as the benefits that adaptation and mitigation measures (for climate impacts) in a given sector have on adaptation and mitigation within the same or in another sector. It was difficult from the limited review of only 25 papers for each adaptation and mitigation measure to identify specific interactions between these measures as they were often not the focus of the paper or the potential benefit for adaptation or mitigation in another sector was not made explicit. It was generally easier to identify synergies within a given sector, as many of the papers were sector focused. It is very important to highlight here that the majority of synergies listed in this section are potential synergies only; often not identified explicitly in the literature, although some explicit examples were found in studies adopting a more multi-disciplinary approach.

In theory, synergies may occur within the same or with different sectors and between sectors, and between:

- adaptation measures (Sections 7.2.1 and 7.2.2), e.g. urban trees used to reduce runoff also reduce the effects of the urban heat island;
- adaptation and mitigation measures (Sections 7.2.3 and 7.2.4), e.g. the practice of returning crop residues in agriculture to improve water use efficiency also improves carbon storage;
- mitigation measures (Section 7.2.5).

An overview of synergies found in this review is given in Table 8. Note that these are both explicit and implicit.

7.2.1 Synergies between adaptation in the same sector

No explicit within sector synergies were identified, but some potential synergies can be proposed. Adaptation measures in the same sector are often aimed at addressing different, but related issues. For example, crop breeding may seek to reduce climate stresses while maintaining/increasing yields or addressing climate-related increases in pests or diseases. The benefits of conservation agriculture techniques include improved crop growth and productivity of the farm system (Munoz *et al.*, 2007) and increased soil bulk density (Tebrügge and During, 1999; Alvaro-Furentes *et al.*, 2008a), making this a viable option for European agriculture in terms of productivity (van den Putte *et al.*, 2010) and could reduce the pressure for increased production.

Synergies within a sector, however, may be complementary or alternative measures for dealing with the same issue. For example, there are a number of ways in which stormwater management can be addressed in urban areas through the use of different types of greenspace such as green roofs (Oberndorfer et al., 2007; Fioretti *et al.*, 2010) and urban trees (Gill *et al.*, 2007) (see Sections 3.5.2, 3.5.7). Also there are a variety of hard and soft engineering options for adapting to rising sea levels and storm surges, which may be able to be combined.

Other possible examples include some of the biodiversity adaptation measures, such as those which are aimed primarily at enhancing the status and condition of habitats and species, thus increasing their size/population numbers and potentially enhancing connectivity. They are mutually compatible and synergistic, except where the requirements of one species of conservation concern are in opposition to those of another.

Table 8: Synergistic interactions between sectors.

	Urban	Coasts	Water	Agriculture	Biodiversity	Forestry
Urban	**Greening** (shade, water, CO_2 storage, avoided emissions)	X	**SUDS/greening:** improves infiltration, evaporation, reduces runoff **RWHS** decentralises management, reduces demand	**Intensification** helps preserve famland	**Intensification** helps preserve open spaces – helping biodiversity to adapt	X
Coasts	X	**Saltmarsh creation** (defence, GHG storage) **Tidal Barrages** (defence, green energy)	X	X	**Saltmarsh creation** provides habitat space	X
Water	**SUDS measures**	X	X	**Flood protection** can have positive impact on agriculture	X	X
Agriculture	X	X	**Intermittent irrigation** reduces water demand **No-tillage** reduces runoff, increases soil moisture, reduces demand	**Returning crop residues** **Water saving irrigation** (water/energy savings, reduced CH_4 emissions from rice paddy)	X	X
Biodiversity	X	X	**Restoring peatlands** improves water regulation and retention, stabilizes water levels	X	**Restoring peatlands** (habitat provision, climate regulation, GHG storage)	**Assisted colonization** e.g. in forests **Artificial regeneration** can accelerate adaptation
Forestry	**Urban forests** reduce mitigation required, provide shading	X	**Increased forest cover** can reduce peak flows	X	**Afforestation** increases habitat area	X

7.2.2 Synergies between adaptation in different sectors

Most of the potential synergies between adaptation measures in different sectors, while being implicitly synergistic were not promoted as such, thus the opportunity for enhancing the adaptation co-benefits was not realised. The various green infrastructure measures employed in urban areas are a good example here, having a range of synergies which may benefit more than one sector. SUDS for example, whilst aiding adaptation for the water sector, have been shown to restore some ecosystem functions in urban areas, such as habitat restoration (e.g. from green roofs), and the replenishment of soil moisture (Spatari *et al.*, 2011). SUDS greening measures and wetland creation have synergies with biodiversity, providing both feeding and habitat areas for birds and insects (e.g. Chance, 2009). Further examples are discussed below for each sector.

<u>Sectors with synergies for biodiversity</u>

A number of sectoral adaptation strategies were found to have potential for synergies with biodiversity. These occur within agriculture, coasts, urban and water sectors; with no adaptation measures in forestry identified as having benefits for biodiversity adaptation. This is interesting in itself, as restoration and afforestation schemes provide habitat space, and can prove to be important corridors for the migration of species. Measures considered in this review to have synergies for biodiversity are those which increase resilience in this sector, by for example the creation and restoration of habitat.

<u>*Agriculture*</u>

A benefit of conservation agriculture for biodiversity was identified in a Hungarian study, in which a higher abundance of seed-eating birds were observed on conservation, rather than conventional-tillage, plots (Field *et al.*, 2007). Many of these birds (yellowhammer: *Emberiza citrinella*, chaffinch: *Fringilla coelebs*, goldfinch: *Carduelis carduelis*, greenfinch: *C. chloris*, lesser redpoll: *C. cabaret*, brambling: *F. montifringilla*, linnet: *C. cannabina*, and reed bunting: *E. schoeniclus*) have been in decline elsewhere in Europe, probably due a lack of food availability resulting from factors such as a switch from spring to autumn sown crops and the use of agro-chemicals (Newton, 2004). In addition, the planting of shade trees to reduce the impact of heat stress in livestock can increase biodiversity (Iglesias *et al.*, 2007) and this may contribute to adaptation.

In China, long-term tillage practices impact on the population density and spatial distribution of biodiversity in agro-ecosystems. Zhu *et al.* (2010) found that conservation tillage will increase the abundance of soil fauna, and Xiong *et al.* (2008) observed that the amount of microorganisms and microbial biomass in the 0-5 cm layer of no-tillage soils were significantly higher than that of the 5-10 cm layer, whereas the differences were not significant in conventional tillage soils.

<u>*Coasts*</u>

Coastal adaptation interventions, such as managed realignment, managed retreat and restoration projects, tend to impact positively on biodiversity via the creation of valuable intertidal habitat (Woodward and Wui, 2001; van Dyke and Wasson, 2005) and possibly increased net biodiversity as a result (Bernhardt and Koch, 2003). The case study of restoration of the Ballona Wetlands, South Carolina, although outside of the study area, is used here to highlight the value of this coastal habitat (Tsihrintzis *et al*., 1996). The restored wetlands had large benefits to biodiversity, one study documenting its use by a number of sensitive and endangered species, including a range of plants, insects, birds and mammals including the Western Harvest Mouse, California Brown Pelican and the California Legless Lizard (Tsihrintzis *et al.*, 1996). In a UK study, Mander *et al.* (2007) reported the created mudflats on the Humber estuary after managed realignment to be highly productive habitats.

As far as individual species are concerned, wetland creation on the Dee Estuary, UK, resulted in the site having an increased carrying capacity for waterfowl, as well as providing a breeding site for rare bird species (Wells and Turpin, 1999). These positive impacts on the biodiversity sector were highlighted in a number of studies; with wetlands being valuable for avian communities, including migratory birds, such as the endangered whooping crane (Hofstede, 2003; Hinkle and Mitsch, 2005; Darnell and Heilman, 2007), waterfowl (Marcus, 2000; Pontee, 2007) and wildfowl (WWF, 2002). A paper discussing the effects of managed realignment at Nigg Bay in Scotland, reported the site to provide an area for foraging and resting at low tide, and roosting at high tide, as well as hosting large numbers of non-breeding waterbirds such as the Common Redshank, and Eurasian Oyster Catcher (Crowther, 2007). The restoration of wetlands for flood storage in the Petite Camargue, southern France acted to increase the biodiversity in the area, including a shift to hydrophyte dominant communities, and increased numbers of fish-eating birds and tree-nesting herons (Mauchamp *et al.*, 2002). Further species are expected to recolonise the area, although these could reduce water quality, and are therefore monitored. Impacts on aquatic species are also numerous, with tidal restoration and improved hydrological connectivity increasing the area of available habitat space (Marcus, 2000; Teal and Weishar, 2005; Pontee *et al.*, 2006; van Proosdij *et al.*, 2010). Restored tidal flow has further been recorded to result in significantly increased nekton density and species richness (Tsihrintzis *et al.*, 1996; Roman *et al.*, 2002).

It is important to note that the aforementioned benefits to the biodiversity sector can be maximised during the design phase of these coastal adaptation schemes. One example of this is the managed realignment scheme at Wallasea, UK, which was designed by DEFRA (Department for the Environment, Food, and Rural Affairs) to provide habitat for a specific assemblage of bird species, and also to provide benefits for fisheries (Dixon *et al.*, 2008). However, perhaps the best example of a project managed with a focus on biodiversity, is a US saltmarsh restoration program at Delaware Bay (Balletto *et al.*, 2005). The Delaware Bay restoration, as part of a biodiversity offsetting project had specific targets to meet and therefore a number of actions were taken to ensure these targets were met, including the installation of fish ladders, so that herring would be able to return to the site. It is important to note that evidence of schemes with benefits to this extent was not found in Europe, although it does highlight their potential. It is also worthy to further mention that as in the example of Wallasea, where the key focus of a scheme is biodiversity, there may be difficulties in gaining the approval of landowners, likely to be adversely affected through the removal or realignment of current defences.

Biodiversity offsetting (Section 16.6) involves conservation activities designed to result in benefits for biodiversity and to compensate for losses resulting from development (Defra, 2011). It is often a requirement of coastal developments and is another example of a cross-sectoral benefit which could possibly be considered to have a synergy with biodiversity adaptation if more than a direct area replacement occurs and is maintained.

The majority of hard-engineering adaptation options have been shown to impact adversely on biodiversity through the promotion of coastal squeeze (Beeftink, 1975; Bozek and Burdick, 2005) (see Section 8.2.1). On the other hand, a small proportion of the literature reveals that seawalls, breakwaters and other low crested structures (LCS) can provide novel habitat for a range of species as a result of the sheltered, shaded conditions (Martin *et al.*, 2005; Glasby *et al.*, 2007; Bulleri and Chapman, 2010). To highlight the potential impact of sea walls, in Australia studies have found these structures to provide artificial habitat similar to that of rocky shores (Chapman and Bulleri, 2003). Some species were identified as being unique to seawalls, and it was found that the structures hosted a different intertidal assemblage to that of natural habitats (Chapman, 2003; Glasby *et al.*, 2007). European studies show that breakwaters can have a positive effect on fish populations by replacing the role played by artificial reefs as a habitat for adult fish, and a nursery ground for juveniles, therefore having the potential to benefit local fisheries (Guidetti *et al.*, 2005; Lamberti *et al.*, 2005; Martin *et al.*, 2005). From this review, it was unclear whether LCS had a net positive effect on biodiversity, with one study finding the structures to host less diverse epibiotic assemblages, providing a habitat which was easier to colonise (Moschella *et al.*, 2005), whereas another reported an increase in diversity, species numbers and species richness of the surrounding area (Martin *et al.*, 2005).

Urban

The majority of urban adaptation-mitigation studies relied on some form of green infrastructure – for example, street trees (Lafortezza *et al.*, 2009; Hall, 2012), green roofs (Bates *et al.*, 2009; Fernandez-Canero and Gonzalez-Redondo, 2010), and urban greenspace (Bowler *et al.*, 2010), with a large amount of literature found during the review associated with this term (e.g. Benedict and McMahon, 2002; Gill *et al.*, 2007). Such measures increase the network of green spaces in urban areas, and provide valuable habitat space, hence, benefiting biodiversity adaptation in what may otherwise be a more hostile environment to many native species. Green roof systems, for example, can host a variety of insects (Coffman and Davis, 2005), and are potentially valuable sites for bee conservation (Tonietto *et al.*, 2011). For birds, green roofs can provide a source of water and food, as well as offering space; protecting both them and their nests from predation (Baumann, 2006).

Urban street trees are another example of an adaptation measure which can help to form part of a city's green infrastructure. This is the case in Budapest, where urban trees have been used extensively, with a high biodiversity of 220 species planted which includes both evergreen and deciduous trees (Hegedüs *et al.*, 2011). The urban tree canopy in London also benefits biodiversity, currently covering an area of 157,000 ha (Tallis *et al.*, 2011). Sustainable urban drainage systems (e.g. vegetated swales and ponds) have also been shown to restore some ecosystem functions in urban areas, through habitat restoration, wetland creation and the replenishment of soil moisture (Spatari *et al.*, 2011).

One further, but rather more implicit example of a synergy here concerns infill and re-development schemes as part of urban intensification (Dixon and Depuis, 2003). Containing urban sprawl is known to preserve open spaces such as farmland and natural landscapes (Ancell and Thompson, 2008; Hayek et al., 2010; Searle, 2010), being viewed in the UK as a means to protect greenbelt areas (Williams, 1999), which could aid biodiversity adaptation.

Water

An important adaptation measure that can aid water management and conservation is floodplain restoration, as it provides 'room for the river'. Floodplain habitats such as freshwater wetlands are carbon stores, yet their number and biodiversity are in decline due to competing interests, such as urbanisation and drainage (van Roon, 2012). Protection and restoration of freshwater wetlands, such as peat bogs, to manage water flows could form part of biodiversity adaptation. They can also help mitigate climate change (when undertaken efficiently and in appropriate environments), and contribute to landscape adaptation as they allow for increased water infiltration and storage compared with other land use types (Mitch and Gosselink, 2000).

This review has found that some adaptation schemes, such as floodplain restoration, allow for biodiversity offsetting, e.g. where hard adaptation measures, such as for flood protection, change or destroy habitat, new habitat land is set aside to compensate for losses (see Section 16.6). Offsetting schemes have explicit synergies with biodiversity, and can happen in existing green spaces, between green spaces (e.g. in urban areas, allowing sustainable paving or letting water soak naturally into the ground), or in new green spaces (e.g. climate change could shift ecosystems northwards). Habitat replacement could happen many years after a form of adaptation (e.g. dam or dike building) has occurred, as it is now recognised that reintroducing a habitat into an area can have multiple benefits to biodiversity and water resources in that immediate area and beyond. In water resource management, there continues to be a careful balance between water security and threats to biodiversity. Offsetting and ecological engineering is chosen on a site-by-site basis, driven by a cause or 'need(s)', the nature of the physical environment, the goodwill and attitude of people at the time and after offsetting occurs, and on the financing system of offsetting schemes (Rohde *et al.* 2006; Vörösmarty *et al.*, 2010). Pre-screening sites to deduce their suitability for restoration justifies the project in terms of costs, inefficiencies, ecological benefits and the chances of the scheme being successful. This also considers the flexibility of adapting restoration to a local area and where appropriate data is available to model the system to anticipate potential impacts. Due to the complex interactions, these can have both synergies and antagonisms for water resource management, thus creating dual challenges.

One adaptation measure to address climate change impacts on water temperature is the planting of vegetation to provide additional shade (Dawson and Haslam, 1983). This provides the dual benefits of additional habitat for wildlife and a carbon sink. Studies of shade have been found to be affected by vegetation height, and also the depth and width of the river. For small streams (< 2 m width), marginal grasses and herbaceous plants can be maintained at optimum levels by livestock grazing, with periodic cutting every 3-5 years to avoid excessive shade (Dawson and Haslam, 1983). For larger streams, similar shade levels (known as 'half-shade') can be gained from larger plants and trees placed at regular intervals. With larger vegetation, the aspect and direction of the sun also becomes important to ensure that too much shade does not occur (Dawson and Haslam, 1983). The presence of a tree canopy layers can be a key instigator in improving soil conditions, and hence in the establishment of other plants that colonise the river. Thus, the effectiveness of vegetation to reduce temperature in rivers evolves over time and, in places, needs to be managed to ensure good water quality and can aid the restoration of riparian habitats (Stockan *et al.*, 2012).

<u>Sectors with synergies for water</u>

Measures taken in various sectors which have synergies for adaptation in the water sector are mainly those which reduce water demand or increase storage capacity. Sectors found in this review to have synergies for the water sector include agriculture, biodiversity, urban and forests.

Agriculture

Relatively few positive interactions were found between adaptation in agriculture and the water sector, those that did being mostly small scale and regionally specific. As far as water quantity is concerned, although it falls outside of the study area, one of the best examples comes from the Great Plains, US. Here, simulated rainfed production of irrigated spring wheat increased significantly under all climate scenarios due to increased precipitation, thus decreasing water demand for irrigation (Tubiello *et al.* 2002). Similarly rain fed production of potatoes could become more competitive than irrigated, although there are studies that indicate the opposite effect, and hence an antagonistic interaction with the water sector (Giannakopoulos, 2009).

In China, agriculture is the biggest water consumer, accounting for 61.3% of total water consumption in 2010 (Ministry of Water Resource, 2012). The average water productivity of China is only 0.61 kg ha^{-1}, much lower than that of developed countries (Wang *et al.*, 2012), and water-saving irrigation here will inevitably exert a marked influence on the water sector. According to recent research, drip irrigation can save 2,250 m^3 of water per ha in Beijing (for vegetable fields; Yang, 2011), and 5,454 m^3 ha^{-1} in Northwest Hebei (Qu *et al.*, 2011).

Conservation tillage practice also has a positive impact on the water sector, leading to a reduction of soil water evaporation, improvement of the soil water storage capacity for rainfall and lesser irrigation demands. Zhang *et al.* (2002) summarized that compared with traditional tillage, conservation tillage can reduce water consumption by 15% and increase water use efficiency by 10%. Furthermore, along the Huaihe River, conservation tillage was shown to reduce water consumption for winter wheat by 15-25% (Wang *et al.*, 2010b).

Biodiversity

The de Doorbraak project in the Netherlands was undertaken with dual goals for the water sector and biodiversity (Box 1; Section 8.2.1). As far as the latter is concerned, the corridors created connect Northeast Twente with the Crest of Salland. This scheme has a number of synergies with the water sector, by creating extra drainage and facilitating a larger water holding capacity, hence, decreasing the risk of both flooding and drought through the creation of various retention areas (WRD, 2011).

Urban

Many urban adaptation strategies adapt areas such as cities to changes in precipitation and water availability. Increases in greenspace and vegetation can improve hydrological performance in the urban area (Armson, 2012). Sustainable urban drainage systems (e.g. greenspace, urban trees, vegetated swales), which often form part of green infrastructure, can reduce runoff volumes and delay peak discharge by improving infiltration and evaporation (e.g. Oberndorfer *et al.*, 2007). They also reduce the likelihood of overflow in combined sewer systems (Semadeni-Davies *et al.*, 2008a), and reduce diffuse pollution in urban watercourses (Scholz *et al.*, 2006; Casal-Campos *et al.*, 2012), whilst being simultaneously able to reduce urban heat island effects as a result of evaporative cooling, shading and albedo effects (Berkooz, 2011; Spatari *et al.*, 2011). Such green infrastructure components therefore diminish the need for adaptation by other sectors.

Urban trees reduce runoff volumes in urban areas by intercepting rainfall (Gill *et al.*, 2007) and return a substantial part of the precipitated moisture back to the atmosphere by evaporation (Rotherham, 2010). A study of the Greater Manchester area found that increasing urban tree cover by 10% would help the city adapt to the increases in precipitation predicted under climate change by intercepting rainfall (Gill *et al.*, 2007). Street trees are also being planted in Sheffield for the purpose of stormwater management, with 120 mature trees planted on a major road corridor by Sheffield County Council and South Yorkshire Forest (Stovin *et al.,* 2008).

To further demonstrate the effectiveness of SUDS measures in reducing runoff, the review found green roofs were associated with reductions in peak flow of 74 ± 20% measured in central and north-western Italy during the autumn and winter seasons (Fioretti *et al.*, 2010). In addition green roofs were measured to delay peak flow by over 2-hours, with the vegetation able to retain an average of 23 ± 31% of precipitation (Fioretti *et al.*, 2010). This potential for stormwater management appears to be comparatively lower in Manchester, where green roof systems were found to reduce runoff from an 18 mm rainfall event by only 17-19.9% (Gill *et al.*, 2007). In Brussels, a modelling study examining the effectiveness of green roof systems as a form of climate change adaptation found that the extensive use of these systems on 10% of the current building stock would reduce runoff in the region by 2.7%, and by 54% on an individual building basis (Mentens *et al.,* 2006).

Factors influencing the stormwater adaptation potential of green roofs include the amount that plants transpire, those with a greater canopy biomass providing a larger total area for gas exchange (Lundholm *et al.*, 2010). In addition, roofs covered in mosses such as *Racomitrium canescens* were found to have a 12-24% higher stormwater retention than vascular or medium-only roofs, for example, being able to hold 47 L m^{-2} without any medium, compared to water storage of 33 L m^{-2} by a roof with a 2.5 cm deep layer of medium (Anderson *et al.*, 2010).

In addition to SUDS and green infrastructure, rainwater harvesting systems installed at the Beddington Zero Energy Development have substantially reduced mains water consumption; that of homes in the BedZED community being almost 60% below the London average (Chance, 2009). Although not explicitly mentioned, this reduces the amount of adaptation required by the water sector, and hence is a synergistic interaction between the urban and water sectors.

Forests

Afforestation can have numerous beneficial impacts on the water sector, including reductions to peak flows and runoff, whereas tree removal has the opposite effect (Trabucco *et al.*, 2008). Typically, once a closer forest canopy has been established, regions in Europe can experience a reduction in peak flow of around 10%-15% (Robinson *et al*., 2003). In areas of low flow, forestry drainage channels can augment baseflows via deeper soil drainage, although these decrease over time due to forest cover and detritus filling the drainage channels (Robinson *et al.*, 2003). In the Spanish mountains, afforestation reduced peak flow in rivers and the sediment size, causing a re-establishment of plants in the river channels and banks (Ortigosa and García-Ruiz, 1995). Through four global case studies, Trabucco *et al.* (2008) found that almost 20% of modelled reforested or afforested land showed little or no change to runoff, whereas another 28% of land demonstrated a moderate impact to reforestation or afforestation. Around 27% of the land modelled had a high impact as runoff decreased by 80%-100%, and this was particularly acute in drier areas, semi-arid tropics and in conversion of dry lands to subsistence agriculture. Although there were significant impacts on local hydrological cycles due to the size of catchment involved, it did not have a significant effect on a regional scale. These reductions in runoff can reduce the impact of intense precipitation events and, hence, reduce the likelihood of flooding. This link was not made explicitly in the above examples, although the studies did examine impacts of forestry adaptation measures on water.

Sectors with synergies for forests

An important related issue in reforestation projects is the balance between natural and artificial regeneration, i.e. to what extent should natural regeneration be used and when to encourage planting of seeds or seedlings, possibly originating from different climatic conditions. The occurrence of frequent natural regeneration is fundamental for continuous natural selection in forest ecosystems, thus maintaining the evolutionary process of forest tree populations. Assisted colonisation is often seen as being particularly appropriate for the conservation and restoration of systems, such as forests, in response to climate change (e.g. Chapin *et al.* 2007; McKenney *et al.* 2009), and attempts at the diversification of tree species can lead to a variety of available habitat types suitable for native species (Lamb 1998; Norton 1998; Hartley 2002). The design of reforestation projects can have a large impact on biodiversity, and could be carried out in such a way as to enhance climate change adaptation opportunities for these species. Artificial regeneration may be needed to complement natural regeneration and, in some cases, to accelerate the adaptation of forest trees to climate change (e.g. by using more southerly provenances). In this way it could reduce the need for adaptation by those concerned with habitat maintenance for its own sake, or for associated forest species.

7.2.3 Synergies between adaptation and mitigation in the same sector

The greatest numbers of explicit synergies recorded were between adaptation and mitigation, whether within or between sectors. For a number of measures there were both within and between sector synergies and in order to avoid the unnecessary division of the synergies associated with a measure, they are discussed together under the category of “synergies in the same sector”.

Agriculture

Explicit synergies within the agriculture sector have been discussed in the wider literature. Howden *et al.* (2007), for example, in the context of discussing adapting agriculture to climate change, suggested that the possibility and costs of implementing both climate mitigation and adaptation is a useful area of study. Rosenzweig and Tubiello (2007) suggest that in many instances mitigation and adaptation strategies in agriculture are synergistic, such as increased irrigation enhancing carbon sequestration. The major synergies in agriculture between adaptation and mitigation are discussed by Smith (2012). They identify that adaptation measures such as (1) reducing soil erosion and the leaching of nitrogen and phosphorus, (2) soil moisture conservation measures, (3) increasing crop rotation diversity by choices of species or varieties, (4) microclimate modifications to reduce temperature extremes and provide shelter, (5) land use change involving abandonment or extensification of existing agricultural land, or (6) the avoidance of the cultivation of new land, will also contribute to mitigation. In general, if properly applied, these measures will reduce GHG emissions by improving nitrogen use efficiencies and soil carbon storage.

This review identified a number of possible synergies. For example, spring sown crops have the potential to reduce N_2O emissions, as they require lower nitrogen inputs than winter sown crops (Olesen *et al.,* 2004). In the case of delayed planting of winter cereals in Denmark, the most efficient way of reducing GHG emissions whilst maintaining yields was seen in a spring cereal based crop rotation with catch crops (Olesen *et al*., 2004).

The conversion to no-tillage management for climate change adaptation carries with it a number of both within and between sector synergies. These include reduced erosion rates, reduced surface runoff, and increased soil moisture availability (Bescansa *et al.*, 2006; Sip *et al.*, 2009; Soane *et al.*, 2012); all of which reduce the amount of additional adaptation required by the agricultural sector to cope with climate change. No-till practices also reduce the lateral loss of agro-chemicals, such as herbicides (Tebrügge and During, 1999), and reduce N leaching from the farm system (Constantin *et al.*, 2010) and could contribute to sustaining carbon storage or sequestration.

A large number of adaptation measures reviewed for China were found to impact positively on mitigation, and some could also contribute to adaptation in the water sector by reducing demand. Water saving irrigation, for example, was found by various researchers to lead to substantial energy savings in agriculture (Guan 2004; Dang *et al.*, 2006; Ma and Feng 2006; Li *et al.*, 2007). Zou *et al.* (2012) found that the 3-year total CO_2 emission reduction for water saving irrigation stands at 34.67 Mt and about 11.56 Mt per year in China. In terms of methane, intermittent irrigation for rice paddy could decrease emissions by 33-93% (Zou *et al.*, 2012).

Returning crop residues, which is regarded as a component of conservation/no-tillage, is proven to increase the water use efficiency and soil carbon storage simultaneously (Wang *et al.*, 2006). Jin *et al.* (2008) observed that the recent increase of soil carbon content in arable land in China resulted from the promotion of returning crop residues and conservation/no-tillage. A national estimation concluded that returning crop residues (50-100%) and no-tillage (50-100%) have a soil carbon storage potential of 23-57 Tg C a^{-1} and 22-43 Tg C a^{-1} respectively (Yan *et al.*, 2007; Wang *et al.*, 2011). Wang *et al.* (2011a) reviewed the carbon sequestration potential of major agricultural practices, the results of which are shown in Table 9, further highlighting the potential for synergies between adaptation and mitigation in the agricultural sector.

Table 9: Estimation of carbon sequestration potentials for major agricultural practices in China.

Agricultural practices	**Carbon sequestration potential (Tg C y^{-1})**	**Literature source(s)**
Mineral fertilizer	94.9	Han *et al.*(2008)
Organic fertilizer	5.5 - 42.2	Han *et al.* (2008); Yu and Li (2009)
Returning crop residue	18.3 - 57.1	Yan *et al.* (2007); Han *et al.* (2008); Wang *et al.* (2009); Yu and Li (2009)
Conservation tillage	2.4 - 4.6	Han *et al.* (2008); Yu and Li (2009)
Conservation tillage (50-100% of arable land)	21.5 - 43.0	Yan *et al.*(2007)

*summarized by Wang *et al.* (2011a)

Biodiversity

Given the carbon content of biomass, any measure that results in its increase will lead to greater carbon sequestration, while adaptation measures which conserve or enhance carbon-dense ecosystems like peatland and forest will similarly contribute to mitigation through carbon storage.

The Restoring Peatlands Project is an example of habitat restoration for substantial areas of degraded peatland in both Belarus and the Ukraine (see www.restoringpeatlands.org). A main aim of the project is the provision of suitable habitat for a number of species, thus helping to conserve diversity under climate change. As far as synergies are concerned, this adaptation measure reduces GHG emissions, hence mitigating climate change, with the Belarus part of the project estimated to sequester a total of 2.9 tons CO_2 equivalent ha^{-1} y^{-1}. In addition, the restored peatlands have other benefits, through regulation of the local micro-climate, improving soil quality and reducing the likelihood of peat fires.

Coasts

A large number of the coastal adaptation measures reviewed impact positively on mitigation, although it is important to note that again these synergies are rarely identified in the literature, with carbon sequestration being considered more of a co-benefit, rather than the reason for implementing a scheme. There is a general consensus that saltmarsh creation, whilst providing a natural coastal defence, is also an effective carbon sink and therefore can contribute towards a reduction in global GHG emissions (Connor 2001; Trulio *et al.*, 2007; Yu and Chmura, 2011). Research has shown that increases in atmospheric CO_2 levels and global warming may accelerate the rate at which marsh systems sequester carbon (Connor *et al.*, 2001; Choi and Wang, 2004), and a review by Trulio *et al.* (2007) demonstrates the ability of saltmarsh restoration to act as a natural defence which will increase in area and height with sea-level rise, therefore creating a highly effective carbon sink.

Studies adopting a cost-benefit approach are among the few which explicitly identify the synergy of saltmarsh as a coastal defence and a reservoir for carbon. Luisetti *et al.* (2011), for example, consider various managed realignment scenarios with the most extreme scenario for the Humber Estuary resulting in 102 km of realigned defences, 34 ha new intertidal habitat and an estimated 3,597.1 t C stored per year as a result of the intertidal habitat created. The study also assessed the most extreme scenario for the Blackwater Estuary with 40 km of realigned defences and a subsequent 639.49 t C stored per year by 2,000 ha of new intertidal habitat. The disparity in the rate of carbon storage and areal extent of intertidal habitat highlights that other factors, such as the amount of agricultural area that will be lost and the overall length of defences removed, also impact on carbon sequestration. Similarly, Shepherd *et al.* (2007) discuss realignment as a coastal adaptation which impacts positively on climate mitigation, with created saltmarsh along the Blackwater Estuary capable of storing 2350-9417 t C per year depending on sedimentation rate.

This review has found that hard-engineering schemes can also impact positively on mitigation. Clark (2006), for example, considered the adaptation intervention of tidal power barrages in the UK and their potential impacts. The study concluded that if enough generators are installed, the barrages would have the cumulative ability to limit local tidal range. Furthermore, they would also impact positively on mitigation by replacing carbon-intensive forms of energy with tidal power, and it is estimated that a barrage on the River Severn would be able to supply 6% of the UK's energy requirements (Clark, 2006).

Forests

Forests and wetlands exhibit the highest capacity for the provision of long-term sequestration of carbon (MEA, 2005) and thus any afforestation or reforestation as part of climate change adaptation will implicitly contribute to mitigation, both through storage in the vegetation and soils. Lal (2004) found that afforestation can enhance soil organic carbon, thus decreasing atmospheric concentrations, although the rate of uptake is variable, related to interactions between climate, soils, tree species and management, and the chemical composition of detritus. Climate change potentially could also enhance forest growth as more nitrogen may be available (i.e. increasing biomass and hence carbon sequestration), and this could compensate for the release of soil carbon in response to warming (Lal, 2005).

Urban

There are several examples of urban adaptation measures contributing to mitigation, without being associated with a particular mitigation measure. Urban adaptation can contribute to mitigation in two ways: (1) through avoided emissions, and (2) through carbon storage. As far as the former is concerned, the climatic regulation and shading effects of urban trees, in addition to reducing runoff, can be associated with decreased emissions stemming from a reduced use of active cooling systems in buildings (Nowak, 1994; McPherson *et al.*, 2005). These avoided emissions will be especially large in regions with high cooling loads, and contribute indirectly to climate change mitigation.

To illustrate the possible extent of this synergy with mitigation, it is necessary to draw on findings from a number of US studies, as the review did not find such links in European studies. The city of Berkeley, for example, has a high number of streets with trees (65%), resulting in substantial energy savings from shading during the summer months, equivalent to around 95 kWh per tree (McPherson *et al.*, 2005). Bigham (2011) also identified the potential for energy savings from urban trees, calculating that increasing the number of urban trees in ten US cities by 10% would result in 5-10% energy savings due to shading and wind blocking effects. As a result it has been suggested that a 25-feet tall tree has the capacity to reduce annual heating and cooling costs of a typical residential dwelling in the US by 8-12% (McPherson and Rowntree, 1993). In accordance with this, an interesting scheme in Fresno-California, US, called the PG&E shade tree program sponsors the buyers of new energy efficient homes to plant trees which shade residential dwellings (McPherson and Rowntree, 1993).

Despite the above moderate energy savings, it is important to note, that as far as the avoided CO_2 emissions are concerned, the offsets associated with the shading and cooling by urban trees in Gainesville and Miami-Dade (above), are much lower than those achieved directly by carbon sequestration; at 0.8% and 0.2% of city-wide CO_2 emissions respectively (Escobedo *et al.*, 2010).

As explained in Section 4.5.1, urban trees also mitigate climate change by sequestering carbon (see Nowak and Crane, 2002; Escobedo *et* al., 2010; Davis *et al*., 2011). Hence, in addition to the avoided emissions discussed above, carbon sequestration is yet another synergy associated with urban trees. The amount of storage is variable, it being estimated that trees in Leicester, UK, store around 231,521 t C, equal to a density of 3.16 kg C m^{-2} over the urban area (Davis *et al.*, 2011). Research suggests that if 10% of the present grassland owned by the city council were planted with trees, an extra 28,402 t C would be sequestered into the current pool (Davis *et al.*, 2011). As to achieving the highest levels of mitigation, a US study found that urban forests consisting of natural pine-oak forests, and stands of highly invasive trees achieved the greatest levels of CO_2 storage (Escobedo *et al.*, 2010). It also found that the direct carbon sequestration by urban trees in Gainesville and Miami-Dade is able to offset 2.6% and 1.6% of city-wide CO_2 emissions respectively, suggesting they are a moderately effective mitigation measure (Escobedo *et al.*, 2010).

Despite the above potential, it is important to highlight that urban trees sequester carbon at less than half the density of natural forest (Nowak and Crane, 2002), and hence in mitigation terms, and in contrast to the above finding, the amount of carbon sequestered by urban trees is often seen as negligible; not of great enough magnitude to achieve local greenhouse gas reduction targets (Pataki *et al.*, 2011). Therefore, although this synergy does exist, the use of urban trees primarily as a form of climate change adaptation provides greater benefit, whilst being able to aid mitigation to some extent through emissions avoidance, and (moderate) carbon storage.

Green roofs, while helping urban areas to adapt to climate change through stormwater management and reducing urban heat island effects (discussed in Sections 3.5.3 and 3.5.7), can also make a small contribution to mitigation in terms of carbon sequestration (Rowe, 2011). The magnitude of carbon storage achieved by green roofs in Michigan was calculated at an average of 162 g C m^{-2} (Getter *et al.*, 2009). The potential of this synergy can be improved by altering the species selection, depth of substrate and its composition, and by improvements in management (Rowe, 2011). It is, however, important to note that the potential for co-benefits to be achieved here is rather time-limited, as over time a green roof system will reach a carbon equilibrium, and no longer function as a sink for carbon (Rowe, 2011).

<u>Water</u>

The effect mitigation has on hydrological regimes and vice versa is complex due to the interactions of biological productivity (and carbon in soils), rates of decomposition and GHG emissions (Falloon and Betts, 2010), but there is a potential for adaptation measures that lead to increased soil water and carbon to increase carbon sequestration. The planting of trees and other vegetation, for example, to provide additional shade is an adaptation to potential increases in temperature, which can also act as a carbon sink (as well as providing an additional habitat for wildlife).

7.2.4 Synergies between mitigation and adaptation in different sectors

Synergies between mitigation and adaptation in contrasting sectors were less numerous than those within the same sector, and perhaps reflects the stand-alone nature of strategies. The results from this review suggest that the potential for synergies from other sectors with biodiversity or agriculture (at least in Europe) have not been acknowledged in the literature. No explicit synergies were identified, although for the biodiversity sector, given the role of biodiversity and the increasing interest in ecosystem services, which include climate regulation (e.g. Balvanera *et al.*, 2006), it is likely that many synergies will be made more explicit.

<u>Urban and biodiversity</u>

Urban greenspace is multifunctional and therefore has the potential to both directly and indirectly affect climate change adaptation and mitigation. For example, green infrastructure components, such as urban trees and green roofs, can be employed under adaptation for instances such as shading to reduce heat stress and reduce the urban heat island effect (Doick and Hutchings, 2013), and also to improve hydrological performance (e.g. Bowler *et al.*, 2010; Armson, 2012); but also under mitigation by providing carbon sequestration (e.g. Davis *et al.*, 2011; Rowe, 2011) and emissions avoidance (McPherson *et al.*, 2005). The potential for these measures to impact on adaptation in other sectors has already been discussed in Section 7.2.2, and will be the same for urban mitigation measures.

Water

There are many drivers of change to water resources and to model these requires full integration, including understanding the uncertainties in the drivers, and their impacts. Synergies between adaptation and mitigation are expected, but at times these can also be unexpected and occur at different spatial scales. Given the long time period required to see significant benefits in mitigation, the full impact of synergies may take decades to be recognised. As mitigation studies mainly rely on modelling and prediction, whereas adaptation studies largely report the present situation, there are limited overlaps between the two in the current literature.

To help the water sector and to relate synergies and integration to environmental, economic, urban and social sectors, the EU policies of the Water Framework Directive and the Water Scarcity and Drought Communication have been developed. These evaluate the supplies and demands for water. Quevauviller (2011) states that climate change is not seen as an anthropogenic pressure in the Water Framework Directive, yet over many decades, scientists recognise that climate change does cause changes to water resources and their impacts in many sectors. Climate change and mitigation can influence many steps of the Water Framework Directive and can exacerbate existing problems. The European White Paper on adapting to climate change helps identify these and considers what adaptation strategies can increase resilience over a wide range of sectors influenced by water management, working within the remit of other frameworks and directives (e.g. the EU Floods Directive).

7.2.5 Synergies between mitigation in the same or different sectors

None were identified and this may be because the types of measures considered, e.g. carbon sequestration, enhanced carbon storage and avoided losses usually are treated as complementary, but exclusive measures.

8. Negative interactions

In contrast to the positive and neutral interactions the above mitigation and adaptation measures were found to have, it is also possible that sectors can be negatively affected. These types of interactions are unwanted, and it is important to assess strategies thoroughly to avoid antagonisms (or conflicts) and trade-offs. Negative interactions can take the form of (1) simple negative cross-sectoral interactions (i.e. any direct negative impacts adaptation and mitigation measures have which do not impact mitigation or adaptation), or (2) antagonisms (i.e. where strategies negate other adaptation or mitigation efforts).

8.1 Simple negative cross-sectoral interactions

As with the simple positive cross-sectoral interactions, simple negative cross-sectoral interactions were found to affect the water sector only. In agriculture, for example, the review found evidence that the lack of soil mixing in no-tillage systems causes greater herbicide concentrations to be present in the run-off water, as pesticides accumulate in the upper soil and are therefore not mixed when soil disturbance is minimal (Stevens and Quinton, 2009). The delayed sowing of winter cereals in Denmark was found to result in reduced N uptake by crops during autumn and winter seasons, leading to higher N leaching (Olesen *et al.*, 2004).

In terms of coastal adaptation and mitigation, one of the adverse impacts of wetland creation is a short-term decline in water quality. There are two reasons for this; the first being an increase in concentrations of heavy metals, and the second an increase in nutrient levels, as saltmarsh acts as a sink (Andrews *et al.*, 2006; Loomis and Craft, 2010). MacLeod *et al.* (1999) reported an increase in contaminant metals at Orplands Farm after managed retreat, which led to a short-term reduction in water quality. However, once the system achieved equilibrium with its surroundings, it began to behave as a sink, storing Pb, Cr, and Cu. Many studies recorded increases in nutrient concentrations post-realignment, which has in turn increased the likelihood of eutrophication (Blackwell *et al.*, 2004; Loomis and Craft, 2010). One study concerning the effects of de-embankment on the island of Langeoog, Germany, reported substantial nutrient fluxes of phosphorus and ammonia over the timescale of a few days (Kolditz *et al.*, 2009). There is also evidence that wetland creation alters the redox potential of the soil (Blackwell *et al.*, 2004; Kolditz *et al.*, 2009; Mazik *et al.*, 2010; Thiere *et al.*, 2011). Managed realignment in Devon led to the creation of a new hydrological regime, rapidly lowering the soil-water table, and causing sediments to become anoxic (Blackwell *et al.*, 2004).

8.2 Antagonisms

These are situations where adaptation and mitigation measures in a given sector impact adversely on adaptation and mitigation measures within the same, or in another sector. These may lead to the need for the consideration of trade-offs (Section 10). As with synergies, almost none of the conflicts explicitly mention the impact of an adaptation/mitigation measure on adaptation or mitigation in the impacted sector. These measures could thus be only inferred to as leading to adaptation or mitigation conflicts.

As with synergies, conflicts (in theory) may be within or between sectors and involve:

- Adaptation and adaptation measures (Section 8.2.1), e.g. large-scale coastal defences preventing ecosystems migrating inland;
- Adaptation and mitigation measures (Section 8.2.2), e.g. intermittent irrigation of rice paddies leading to higher N_2O emissions;
- Mitigation and mitigation measures (Section 8.2.3).

An overview of the antagonistic interactions found is given below in Table 10.

8.2.1 Antagonisms between adaptation measures

There are a number of implicit examples of antagonisms, especially in relation to the negative impacts of an adaptation measure on biodiversity, since two of the UK climate change adaptation principles for biodiversity are to conserve and restore existing biodiversity and reduce sources of harm not linked to climate (Smithers *et al.*, 2008). For example, green roofs may have negative effects on native species, as may no-tillage systems, forestry plantings and operations, while some coastal hard-engineering could prevent coastal ecosystems migrating inland in response to sea-level rise .

Table 10: Antagonisms between mitigation and adaptation.

	Urban	Coasts	Water	Agriculture	Biodiversity	Forestry
Urban	**Urban Intensification** paradox	X	X	X	X	X
Coasts	X	**Storm-surge barriers** **Wetland creation**	X	**Restoration** – loss of agricultural land could increase need for intensification of agriculture	**Hard engineering** – prevents some coastal systems migrating inland **Storm-surge barriers** – removal/degradation of habitat **Restored tidal flow** – loss of protected areas, e.g. national park/reserves	X
Water	X	X	X	X	**Building dams** can prevent movement of organisms/cause habitat loss	X
Agriculture	X	X	**Earlier sowing dates** increase demand for water irrigation	**Intermittent irrigation** **Conservation agriculture** **Soil-C management**	X	X
Biodiversity	X	X	X	X	X	X
Forestry	X	X	**Afforestation** can increase water demand Lead to lower base flow/promote less groundwater recharge Deplete flows during drought periods	X	Tillage, ploughing and scarification for **afforest/reforestation** may negatively affect some organisms **Plantations** have less habitat diversity and complexity, reducing ability of some species to colonise **Clear-cutting** increases long-term forest health, but can lead to habitat loss and fragmentation	**Thinning** – reduced carbon storage

There are also possible antagonisms between some of the sectors and water. For example, afforestation on new land can increase water demand, as can crop irrigation. Increasing water supply is necessary to meet demands of urbanisation or improve economic activity (Vörösmarty *et al.*, 2010) and all these changes can impact biodiversity, especially river and wetland species, habitats, and their adaptation to climate change. In urban areas future water pressures also may result in conflict over use of water for irrigation, so it would be important to develop sustainable irrigation measures for greenspace, for example, by rainwater harvesting, the re-use of greywater and floodwater storage, to ensure that they continue to regulate urban climate (Gill *et al.*, 2007). While there are various benefits associated with conservation agriculture practice, there also exist a number of conflicts (Soane *et al.*, 2012), particularly associated with its environmental impacts, including the impact of increased herbicide use on water quality.

Other conflicts may arise with the different land use requirements of adaptation measures. Biodiversity, for example, would require additional land for networks and thus could be in competition with demands from the agriculture and forestry sectors (BMU, 2008). Such conflicts between competing land uses is not new, although climate change adds new dimensions to it and possible solutions are discussed in Section 16.6.

Sectors with antagonisms on biodiversity

Antagonisms for the biodiversity sector were the most numerous found in this review. Evidence of this was found in adaptation strategies for coastal, agricultural, forest and water sectors.

Coasts

The majority of hard-engineering adaptation options have been shown to impact adversely on biodiversity through the promotion of coastal squeeze (Beeftink, 1975; Bozek and Burdick, 2005). Storm surge barriers, such as those created as part of the Netherland's Delta Plan, are a prime example of how coastal adaptation to climate change can have antagonisms for biodiversity. These impacts are shown and discussed in Box 1. Additionally, the construction of embankments in the Wadden Sea area during the last 500 years have resulted in the loss of oyster beds and a significant decline in habitat diversity, with saltmarsh having previously provided an important habitat for migratory species (Reise, 1998). In a number of studies, the construction of LCS was shown to result in an increase in the abundance of algae, which hindered the ability of species to settle and reproduce on the structures (Blockley and Chapman, 2005; Lamberti *et al.*, 2005; Martin *et al.*, 2005).

The Delta project, Netherlands

The Delta Project led to the closure of numerous estuaries along the Dutch coast, and has had the desired effect with respect to reinforcement of the coastline, and protection against flooding (Wolff, 1992). In contrast, the closure of estuaries has led to a subsequent reduction of 120 km^2 in tidal area on Eastern Scheldt, causing many intertidal zones to dry out (Elgershuizen, 1981; Saeijs and Stortelder, 1982; Smits *et al.*, 2006). The new hydrodynamic regime has resulted in widespread erosion, with a reported loss of 120 million m^3 of sediment from the Oosterschelde tidal basin, and a doubling in the rate of cliff retreat has occurred since the completion of the project (Louters *et al.*, 1998). Research shows that the Delta Project will cause the future loss of all tidal flats in the area (Smits *et al.*, 2006), and therefore could be seen as an example of maladaptation, with hard-engineering damaging natural coastal defences, and increasing rates of erosion as a result of a progressively more unnatural regime.

Biodiversity

Perhaps due to the adverse nature of impacts on biodiversity, the majority of studies concerning the Delta Project focused on the link between the coast and biodiversity. In the Eastern Scheldt, high erosion rates resulted in the loss of 170 km^2 of intertidal habitat (Schekkerman *et al.*, 1994), and in addition, Wolff (1992) reported that 70% of marshes along the estuary had lost their character since completion of the project. The combined effects of these impacts are reductions in, and the disappearance of many native estuarine species dependent on the intertidal zone (Schekkerman *et al.*, 1994; Smits *et al.*, 2006). Saeijs and Stortelder (1982) quantified the environmental impact after the closure of the Grevelingen Estuary, reporting the reduced tidal influence to have altered plant and animal communities, resulting in the disappearance of 80% of crab and lobster species, and loss of fish species which were no-longer able to migrate between fresh and seawater habitats. This has impacted adversely, causing considerable damage to local fisheries industries, the area having previously hosted a rich aquatic system (Wolff, 1992). In addition, the decline in native species has allowed a number of invasive and exotic species to establish since construction was completed (Jong and Kogel, 1985).

Water

The barrages removed the tidal influence from a number of estuaries, with the area behind the dams being turned into stagnant lakes, and seawater experiencing desalination (Noordwijk-Puijk *et al.*, 1979; Elgershuizen, 1981; Saeijs and Stortelder, 1982; Jong and Kogel, 1985; Wolff, 1992; Smits *et al.*, 2006). Stagnant water in the lake behind the dam at Grevelingen was found to contain high nutrient loadings, which in conjunction with a long residence time could cause eutrophication of the system (Elgershuizen, 1981; Saeijs and Stortelder, 1982).

One positive impact on the water sector was that the decreased tidal velocity in the estuaries improved water clarity, which in turn could lead to increase in primary productivity by phytoplankton (Elgershuizen, 1981).

Urban and socio-economic

A study by Smits *et al.* (2006) highlights the fact that although the Delta Project has decreased the likelihood of coastal flooding, the perceived increase in safety has led to further growth both in terms of population and the economic value of the area, resulting in a much higher potential damage. In contrast, a study by Elgershuizen (1981) identified the barriers to have the potential for numerous socio-economic benefits, depending on how actively the barrier was managed. These included benefits to shipping; fisheries, with higher water levels over the winter months protecting shellfish from the negative effects of frost; protection against oil-spill events; and the provision of area suitable for water tourism related activities such as sailing.

Box 1: Case study of the Delta Plan which illustrates the interactions between coastal adaptation and other sectors.

This review has also found several studies to suggest that soft-engineering schemes, which are generally considered to have a low environmental impact, can have adverse consequences on biodiversity (Nelson, 1993; de Ruig, 1998; Bishop *et al.*, 2006; Peterson *et al.*, 2006; Jackson *et al.*, 2010). A review by Speybroeck *et al.* (2006) discusses a range of these impacts, which include the compaction of the sand, damage to habitats and plant communities, and the destruction of dune vegetation. Studies from the US, although outside the study area, provided the best examples of the potential antagonisms on biodiversity from coastal adaptation. Bishop *et al.* (2006) assessed that changes to sediment size and density as a result of nourishment altered the assemblages of benthic invertebrates (Bishop *et al.*, 2006). In addition, two studies report the deposition of sand to cause the burial and suffocation of some species, resulting in short-term mortality (Bishop *et al.*, 2006; Peterson *et al.*, 2006). It is important to note the scale of these impacts, which in North Carolina were not only local, but found to affect an area over one kilometre away from the site. Other impacts associated with the compaction of sand after nourishment include a decrease in the area of suitable foraging habitat, with a 14-29% reduction at one site in Carolina (Peterson *et al.*, 2006). Another study from the US showed that beach replenishment had caused the mole crab, which was the dominant swash-zone species, to abandon the site in the short-term (Nelson, 1993). Both of the above studies record a decrease in use of the site by shorebirds after a reduction in the amount of prey available post beach nourishment (Nelson, 1993; Peterson and Bishop, 2005; Peterson *et al.*, 2006). It is therefore clear from this review, that even soft engineering approaches have the potential to antagonise adaptation in the biodiversity sector by removing suitable habitat space and, hence, decreasing the resilience of a number of species under climate change.

Agriculture

There is a strong and clear interaction between crop and livestock breeding and biodiversity, as while it is estimated that there are about 50,000 edible plants, less than 250 are used and 15 crops supply 90% of the calories in human diets, with three (wheat, maize and rice) providing 60% (Ceccarelli *et al.*, 2010). Breeding has been important in increasing yields in all three crops, but it has been at the expense of genetic diversity and, hence, antagonises the ability of biodiversity to adapt to climate change. Natural variations are critical in providing traits not just for increased yield, but also for resistance to disease and pest, drought tolerance, nutritional quality, etc. and are fundamental to conventional and newer genome-assisted breeding approaches (Weckwerth, 2011). It has been suggested that conventional agriculture and plant breeding could lead to the extinction of diverse cultivars and non-domesticated plants (Mendum and Glenna 2010), with Gepts (2006) claiming that modern industrial agriculture is the single biggest threat to biodiversity. Organic agriculture, however, is even more dependent on locally adapted traditional genotypes or landraces to cope with large genotype-environment interactions (Wolfe *et al.* 2008). Climate change may exacerbate the crisis as the lack of genetic diversity hinders adaptation (Ceccarelli *et al.* 2010). Increases in monoculture biofuel production may also lead to greater biodiversity loss, including genetic losses, and the displacement of locally adapted varieties important for climate change adaptation (Sarker and Erskine 2006; Rodriguez *et al.* 2008). Increasing agricultural production through extensification can also conflict with biodiversity, as it increases competition with other land uses, however, the use of algae for biofuels could diminish such competition for arable land, and lead to other environmental benefits, such as the avoidance of soil erosion (Weckwerth, 2011).

The use of water for irrigation can also compromise biodiversity, including endangered species and nature reserves (Adams and Cho, 1998; Hellegers *et al.*, 2001; Iglesias *et al.*, 2011). Wetland habitats are likely to be particularly affected by agricultural water abstraction from ground or surface waters and an investigation of 13 RAMSAR sites in Greece found that irrigation was the most negative action affecting wetland functions and values, followed by cropland expansion and overgrazing (Gerakis and Kalburtji, 1998). Conversely, Voldseth *et al.* (2009) showed that water levels in wetlands could be increased by changing surrounding land use from unmanaged to managed grasslands or cultivated crops. These changes could lead to water levels under a 2°C rise in temperature, both with and without a 10% increase in precipitation, being higher than unmanaged grassland under historical climate conditions (Voldseth *et al*., 2009). This was sustained by managed grasslands under a 4°C rise in temperature with a 10% increase in precipitation and it also reduced the proportion of years in which the wetland dried out by nearly 40% under the three climate change scenarios. Converting the unmanaged grassland to row crops could, however, have the undesirable effect of increased sedimentation and pollution in the wetland (Voldseth *et al.*, 2009).

Forests

The use of chemical control methods for pests and diseases as a form of adaptation to future conditions can have negative impacts on biodiversity, provoking side effects on non-target organisms. For example, according to Raulund-Rasmussen *et al.* (2011) intensive pesticide use negatively affects species that are abundant in forests soils and therefore essential for good soil structure. Insecticides may affect non-target organism, e.g. deltamethrin, which is toxic to fish (Pimpao *et al.* 2007; Berry *et al.*, 2008b). Finally, fertilisation affects the soil quality negatively (Raulund-Rasmussen *et al.*, 2011).

The logging technique of clear cutting can contribute to long-term forest health, although it has the potential for both positive and negative effects on biodiversity. Specifically, for the case of species adapted to old forest environments with small scale disturbances and a long continuity of tree cover, clear-cutting can lead to habitat loss and fragmentation. Nevertheless, clear-cutting can be combined with bio-fuel harvest, decreasing the structural diversity at the site and reducing the ability for some ground living species to survive the open biotope succession phase (Åström *et al.*, 2005; Raulund-Rasmussen *et al.*, 2011).

Water

Vörösmarty *et al.* (2010) found that methods to increase water security (e.g. building dams to store and control water) although having little immediate adverse impact on humans, can have significant adverse impacts on biodiversity in that immediate area or beyond. This includes changes in flow regimes, the movement of organisms, fisheries and habitat loss.

Sectors with antagonisms on water

Agriculture was the only sector identified in this review where measures were found to antagonise adaptation in the water sector through increased demand. It was recognized in the literature that the use of water for irrigation will lead to reduced water availability. This is known to affect river flows and lake levels, as globally only about 25% of the irrigation water withdrawn is taken up by crops, 19% is lost and about 56% is available for subsequent use (Sauer *et al.*, 2010). As far as crop type is concerned, the only cross-sectoral interaction identified for growing spring versus winter crops was with water. The adaptation measure of earlier sowing dates and the use of longer growing cycle cultivars would both require additional water for irrigation, although they have the potential to reduce the negative impacts of climate change by allowing crops to escape higher temperature and water stress (Giannakopoulos, 2009).

Sectors with antagonisms on agriculture

The loss of agricultural land is commonly a direct result of many dynamic coastal adaptation interventions discussed in this review (Gardiner *et al.*, 2007; Pontee, 2007; Marquiegui and Aguirrezabalaga, 2009; Blackwell *et al.*, 2010; Mazik *et al.*, 2010). Managed realignment, for example, included the restoration of tidal flow to 21 ha of arable land at Tollesbury (Wolters *et al.*, 2005b; Garbutt *et al.*, 2006; Reading *et al.*, 2008), and the inundation of agricultural fields and grazing meadow at Orplands (Emmerson *et al.*, 1997). In the UK, depending on the future policy scenario, it was found that in the most extreme case, future managed realignment on the Humber Estuary would result in a loss of 7,000 ha of agricultural land (Luisetti *et al.*, 2011). Although not implied as such in the literature, this loss of land will reduce the ability of agriculture to meet future demand, and hence has antagonisms with adaptation in this sector

8.2.2 Antagonisms between mitigation and adaptation

Adaptation antagonising mitigation efforts

Agriculture

The review of agriculture in China found many instances of agricultural adaptation strategies impacting negatively on mitigation and although some field and micro data have been found, there is no systematic national evaluation, neither qualitative nor quantitative, of these in China. A recent study indicated that reservoirs, which are an important part of agricultural adaptation infrastructure in China, were net emitters of GHGs due to the decomposition of flooded vegetation and soil organic matter. There are three fates for the GHGs produced: direct flux at the air–water interface, turbulent exchange (e.g. in hydroelectric turbines), and spillway downstream of the reservoir (Lima *et al.* 2008).

Intermittent irrigation for rice paddies, which is widely used in China to save water and increase yields under climate change, will create an environment benefiting the production of N_2O and thus result in higher GHG emissions (Akiyama *et al.*, 2005; Zou *et al.*, 2007).

Experiments in Hunan, for example, indicated that non-flooding irrigation could lead to higher denitrification in the 0-10 cm topsoil layer and thus higher N_2O emissions (0.75-2.5 times greater) compared with traditional flooding irrigation methods (Xu *et al.*, 2012).

In Europe, conservation agriculture practices also could lead to increases in N_2O emissions (Ball *et al.*, 1999; Smith *et al.* 2000a; 2001; King *et al.*, 2004; Carlton *et al.*, 2012), with evidence of increases in the nitrogen content of the surface soils in no-till systems from residue and fertiliser applications (Stevens and Quinton, 2009). In terms of adaptation measures for heat stress in ruminants, it is generally faster to improve welfare, production and reproduction performances by altering the environment (West and Marland, 2003; Mader *et al.*, 2006), but intense environmental modification, such as air conditioning, could not only be too expensive, but also the increased energy use would contribute to further climate change.

Coasts

Similarly, some coastal forms of adaptation, such as the construction of tidal and storm-sure barriers in the Netherlands, can impact negatively on mitigation. Here, storm-surge barriers have removed the tidal factor, degraded intertidal habitat and caused the subsequent loss of a carbon sink (Saeijs and Stortelder, 1982; Schekkerman *et al.*, 1994). However, these antagonisms are not explicitly mentioned within the literature, with studies only going as far as to mention the loss of saltmarsh and relating secondary impacts to biodiversity, rather than carbon storage.

Tidal barriers are also capable of impacting on local climates, where for example a number of stagnant lakes have formed barriers constructed as part of the Delta Project (Noordwijk-Puijk *et al.*, 1979; Saeijs and Stortelder, 1982; Wolff, 1992; Smits *et al.*, 2006). These have a high heat capacity and as a result, the lakes could cause summer water temperatures to increase by 1-3°C, and during long periods of stagnation could increase temperatures at a local level (Elgershuizen, 1981). This could be seen as maladaptation as it would then require other (local) adaptation measures and could cause existing adaptation efforts to become ineffective.

Despite the carbon storage benefits from saltmarsh creation, wetlands are also known to be sources of the two potent GHGs; methane (CH_4), and nitrous oxide (N_2O) (Ding *et al.*, 2004; Hansen, 2009; Liikanen *et al.*, 2009; Loomis and Craft, 2010; Moseman-Valiterra *et al.*, 2011; Thiere *et al.*, 2011). Studies differ on the magnitude of flux of these gases, with a study by Bartlett *et al.* (1985) calculating the global annual CH_4 emissions from saltmarsh to be 0.34×10^{12} g CH_4, and other studies reporting CH_4 emissions to be negligible and significantly lower than that of other wetlands (Ding *et al.*, 2004; Trulio *et al.*, 2007; Thiere *et al.*, 2011). As far as the release of nitrous oxide is concerned, Blackwell *et al.* (2010) state that if managed realignment is implemented on a global scale, the production of N_2O would be so great in the mid-term that it would result in a positive feedback effect until sites are fully developed into natural saltmarsh. Similarly, a study by Andrews *et al.* (2006) reports that in the short-term, microbial reactions associated with initial high denitrification in created marsh result in at least a 50% decrease in the benefits from future carbon storage. This concept was also highlighted by Hansen (2009), with CH_4 and N_2O release from created saltmarsh having the potential to counteract the carbon storage benefits. Furthermore, as the majority of schemes in this review involved some form of wetland creation, it is important to note that climate change and marsh degradation could drive these systems to become net GHG sources (Burkett and Kusler 2000; Moseman-Valiterra *et al.*, 2011).

Finally, Luisetti *et al.* (2011) identify an interesting antagonism in that the loss of agricultural land as a result of managed realignment may impact adversely on food security, reducing the ability to cater for future increasing food demands and potentially increasing the need for greater adaptation in this sector. Similarly, they note that although purchasing agricultural land for such schemes comes at relatively low cost, this could rise substantially in the future with population increases leading to increased requirement for food production.

Forest

The literature search found evidence of only one antagonism on mitigation resulting from adaptation in forests, which was associated with the adaptation strategy of thinning. Law and Harmon (2011) stated that this practice is in direct conflict with carbon sequestration goals, because it results in a net emission of CO_2 to the atmosphere.

Mitigation antagonising adaptation efforts

Very few explicit adverse impacts were identified for mitigation measures on adaptation, these were found to occur in agriculture, urban areas and forestry only. Rosenzweig and Tubiello (2007), while recognising the importance of both adaptation and mitigation in dealing with climate change impacts in agriculture, suggest that mitigation measures, such as less intensive production systems and some soil carbon management practices, may compete with adaptations in local agricultural practices aimed at maintaining production and income.

In urban areas, intensification employed as a mitigation measure to reduce emissions could exacerbate existing urban heat island effects (Williams, 1999), and it is known that dense urban areas have higher runoff speeds than low-density suburban areas (Dodson, 2010), which would increase the need for adaptation by the water sector.

As far as forestry is concerned, plantations can lead to significant reduction of agricultural land area, the promotion of farming practices with significant environmental burdens, the conversion of land for cropland expansion elsewhere, and to consequential increased imports of agricultural products (McCarl and Schneider, 2001). Furthermore, the price of farm and agricultural lands is expected to increase due to the lower availability affecting the economic viability of agro-enterprises (MEA, 2005).

Forest plantations can also affect the biodiversity sector in a negative way, especially where they replace biologically rich native grassland or wetland habitats (Nabuurs et al., 2007; Wagner, *et al.*, 2006). According to Brockerhoff *et al.* (2007) plantation forests usually host less habitat diversity and complexity, the possible consequence being that plants and animals which are old forest specialists may not be able to colonise or reproduce in plantations with comparatively short rotations. These rotations not only affect biodiversity (Berry *et al.*, 2008b), but have severe impacts on soil quality (Raulund-Rasmussen *et al.*, 2011). This review therefore found that plantations antagonise the ability of biodiversity to adapt to climate change via reducing the quality of habitat.

Similar antagonisms on biodiversity were found to result from afforestation and reforestation due to the main site preparation methods, including tillage, ploughing and scarification, which can negatively affect some organisms. Bellocq *et al.* (2001) claimed that arthropod diversity declined with increasing post-harvest site disturbance, especially collembolans and mites – which is important for soil fertilility by making adventitious pore structure. Drainage of wet habitats, such as peatland, fens and swamps, for forests has, in the past, led to loss of wetland biodiversity. Raulund-Rasmussen *et al.* (2011) stated that for some species with limited dispersal abilities, the construction of roads, tracks, and other infrastructure within forests may act as barriers eliminating or limiting migration. Soil compaction can occur as the result of off-road driving in the forest, associated with harvesting, and on forest roads and skid trails. Adverse impacts on the root environment are also possible, as compaction means that roots have difficulties of extending during dry summers and during wet winters because of lack of oxygen, leading to lower production rates (Raulund-Rasmussen *et al.*, 2011).

Trees and forests can have a significant impact on the hydrological cycle at various scales and this must be examined thoroughly when undertaking forestry (e.g. Farley *et al.*, 2005; FAO, 2012) and planning effective carbon sequestration measures. Concerns about large scale afforestation firstly include enhanced evaporation loss (compared with crops, so it could result in drier conditions), and increased water use, particularly for coniferous forests and eucalyptus (Robinson *et al.*, 2003). Secondly, there are concerns that forestry can change flows, or can deplete or enhance low flows during drought periods (McCulloch and Robinson, 1993). For instance, if agriculture or crop land was converted to forest, the forested area could exhibit increased actual evapotranspiration and/or decreased runoff. Drainage of peaty soils by open ditches or furrows can increase peak flows and shorten the time for a maximum river height to occur. For example, a site which was drained and afforested with conifers in Coalburn, England resulted in increased peak flows of around 15% for several decades after the event (Robinson *et al.*, 2003). Jackson *et al.* (2005) studied the effect of large scale afforestation of grasslands, finding this could reduce water flow into other ecosystems and rivers, as well as affecting the aquifer layer and recharge, and leading to substantial losses in stream flow. Particularly in dryland areas, plantation species may utilize more water than the natural vegetation, resulting in less recharge of groundwater and reducing stream flow available to other users (Jackson *et al.*, 2008).

8.2.3 Antagonisms between mitigation and mitigation

Almost no examples were found of antagonisms between mitigation measures, although there were several examples of trade-offs resulting from measures increasing emissions of other GHGs (Section 10). One example from the literature examined the mixed effects on GHG emissions from an agricultural created wetland for nitrogen farming (mitigation measure) in Sweden. The authors report that the wetland creation can lead to anoxic conditions and reduced redox potential, making such created systems more likely to emit methane and, hence, requiring further need for mitigation (Thiere *et al*., 2011). Research also shows that higher summertime temperatures will increase the environmental risk for CH_4 emissions from wetlands (Thiere *et al.*, 2011), implying the need for careful assessment of the full range of impacts from this mitigation action.

Taking urban intensification as another example, this measure can have several benefits, however, increased density is also related to greater concentrations of traffic, which can adversely impact local environmental quality (Williams, 1999), and congestion in the locality of the intensified area can increase fuel consumption leading to increased emissions (Melia, 2011). This conflict is known as the ‘paradox of intensification’ (Melia, 2011).

In the water sector, challenges will arise from different mitigation policies. For example, in the UK there is a commitment to produce 15% of all energy supplies from renewable source,s such as bio-fuels and onshore wind turbines, by 2020. Also, the effects of any changing land use and subsequent soil disturbance should be considered and planned for to ensure that soil carbon is suitably stored (Ostle *et al.,* 2009).

9. Conditional impacts

The majority of impacts discussed above, while being primarily positive or negative for another sector, will have some dependence on factors such as location, the manner of implementation and associated management strategies. This was shown quite clearly in the analysis of adaptation and mitigation measures on biodiversity (Berry, 2009a). Nevertheless, a few authors from this review did explicitly add caveats to their findings, or show how the impact could vary depending on circumstances.

One explicit example is the impact of tillage practice on biodiversity. A Hungarian study identified a benefit to biodiversity, in which a higher abundance of seed-eating birds was observed on conservation rather than conventional-tillage plots (Field *et al.*, 2007). However, this benefit existed only during mild winter seasons; as otherwise snow covered the fields, and the birds could not access the seeds (Field *et al.*, 2007).

Also in urban areas, despite the potential for green roofs to provide suitable habitat space, the challenging climatic conditions and location can restrict the use of these sites by a number of species, especially native ones (Brenneisen, 2006). It has been suggested that the suitability of green roofs for birds is highly dependent on the type of green roof, its design, vegetation type and maintenance (Fernandez-Canero and Gonzalez-Redondo, 2010).

In agriculture available moisture affects soil organic carbon (SOC), and while the interrelationship is poorly understood it is thought that increasing cropland irrigation could decrease SOC water storage if NPP was unchanged (Falloon and Betts, 2010). If the potential increase in NPP is taken into account it is more likely that SOC will increase, leading to greater soil water holding capacity, thus possibly reducing the need for irrigation, it could also decrease soil nutrient losses and help to regulate runoff. In northern Europe, increased water abstraction for irrigation and agriculture may decrease the overall supply, but result in increased net primary production, carbon input to soils, above ground carbon storage and soil carbon decomposition as the soils are wetter. Where water supply decreases (e.g. in southern Europe), the opposite may happen. Thus, more effective mitigation measures are required for southern as opposed to northern Europe. The effect mitigation has on hydrological regimes and vice versa is more complex and interactive due to production (and carbon in soils), rates of decomposition and greenhouse gas emissions (Falloon and Betts, 2010).

10. Trade-offs

Trade-offs may have to be made within different adaptation and mitigation objectives and between adaptation and mitigation. Trade-offs also may occur where adaptation and mitigation measures have negative impacts and there were more examples of this (Section 8). In both cases these may be within the sector of interest, in which case they were often more explicit, or intersect with others.

10.1 Agriculture

Given the competition for water and existing conflicts (e.g. between irrigation and public water supply and environmental protection; Daccache *et al.*, 2012), trade-offs are inevitable. Some of these have already been identified as part of cross-sectoral interactions and choices will have to be made, for example, between maintaining water levels for biodiversity and agriculture; switching to irrigation for future potato production in England and Wales and public supply (Daccache *et al.*, 2012); using deficit irrigation while accepting a reduction in yields (Mushtaq and Moghaddasi, 2011).

It is estimated that 30% of the UK wheat acreage is planted on drought-prone land, such that 10% of potential production is lost annually because the moisture available to the crop is insufficient at some point during growth (Foulkes *et al.*, 2007). Irrigation is often an adaptation response to drought or water stress and experiments have shown that irrigation can increase the yield of winter wheat by 17–55% (see Table 1 in Whalley *et al.*, 2006). A comparison of 66 UK winter wheat lines showed a strong correlation between yield under irrigated and rainfed conditions (Dodd *et al.*, 2011). This suggests that, under UK conditions, selection for high-yielding varieties quite often produces varieties which also do well under water-limited conditions. However, varieties that have high yields under optimum conditions, but poor performance under stressed conditions should be identified and culled to ensure effective adaptation. Dodd *et al.* (2011) suggest that the use of varieties that maintain growth and yield as the soil dries could help to avoid half of the yield losses attributed to water deficit. This would boost yields by 5% and thus avoid trade-offs between yield and water usage, as well as avoiding possible demands being made on water resources.

Adaptation through the introduction of a new gene into a cereal genome can significantly alter end-use quality or change tolerance to stresses, although so far there is no convincing evidence that such an introduction leads to increased yields or tolerance to a wide or variable range of stresses (Araus *et al.* 2002). Research suggests that there is a need to better understand the genotype x environment interactions and to identify and locate gene sequences controlling agronomically important traits, thus avoiding trade-offs. It has been noted that cattle bred for improved productivity are more susceptible to heat stress (Nienaber and Hahn, 2007). For example, increasing daily milk yield from 35 to 45 L is thought to increase sensitivity to thermal stress and reduces the ‘threshold temperature’ by 5.8^{o}C (Berman, 2005).

Two further trade-offs between mitigation options in agriculture were found, in which potential GHG reductions are obstructed by increases in others. Firstly, as mentioned in Section 4.1.1, CH_4 emissions from manure stores could be reduced substantially if their temperature were reduced. However, in order to gain the maximum CH_4 emissions reduction, further energy is required to cool the stores, with associated increased emissions from cooling purposes as a result (Dalgaard *et al*., 2011). This presents a trade-off between achieving maximum CH_4 emissions reductions, and the extra energy and emissions which would be needed to facilitate them. Another emissions trade-off is identified for the adoption of no-tillage practices. In this case and under certain conditions, such as wet poorly drained soils, a trade-off could be created whereby high N_2O emissions counteract the carbon storage benefits in terms of global warming potential (Smith *et al*., 2000a; Smith *et al*., 2001; Carlton *et al*., 2012). Such a trade-off is not expected to occur under minimal tillage, and hence, in some cases the adoption of no-tillage as a mitigation strategy may in fact increase GHG emissions (King *et al*., 2004).

10.2 Coasts

Trade-offs mentioned in the literature for coasts tend to be related to managed realignment and retreat, where the trade-off is between the preservation of current primary habitat or land, and the creation of a potentially more valuable area (e.g. through biodiversity offsetting) and a sustainable coastal defence. These problems arise when the area designated for restored tidal flow is protected, for example, a coastal nature reserve, SSSI site, National Park or RAMSAR site. This is a recurrent issue, with one example of managed realignment on the

Humber Estuary leading to the direct loss of 26 ha of intertidal mudflat habitat in an SSSI (Pontee *et al.,* 2006).

Similarly, managed retreat at the Orplands Farm site in Essex led to the loss of grazing marsh in an SSSI, although the overall result of the scheme was an improvement in estuary habitat, with the re-establishment of rare saltmarsh (Dixon *et al.*, 1998). The managed retreat policy being adopted in the Cley Marshes Nature Reserve, Norfolk is another example (Klein and Bateman, 1998). The policy was decided by MAFF to achieve nature restoration, improve economic efficiency of the site, and resilience to extreme coastal events. In this case, the implementation of the adaptation measure caused high amounts of damage, degrading the unique freshwater habitat of the Marshes. Coastal measures can also affect the urban sector, commonly due to the impact of reduced defences on existing coastal settlements. For example, Dixon *et al.* (1998) identified that reduced defences at the Abbotts Hall site led to increased tidal flushing, increasing the likelihood of damage to constructed assets. Similarly, Bakker *et al.* (2002) report the strong tidal currents after de-embankment to have led to the erosion of a small road. One further example of a trade-off found in the literature is at Corton Village, UK, where proposed managed realignment would, on one hand, make the cliff-top village community extremely vulnerable by removing current coastal defences employed as part of the 'hold the line' policy, but on the other, the erosion of cliffs in part of the SSSI would improve its geological value, and is a more appropriate financial solution than the unsustainable maintenance of existing defences (McFadden, 2008). Such trade-offs require careful consideration, and the cooperation of a number of actors.

10.3 Forest

Tabucco *et al.* (2008) state that much attention is given to the international opportunities associated with carbon management, but less to the resulting trade-offs associated with sequestration schemes. For instance, where land use is changed (e.g. from crops to forestry), this can result in changes to the water cycle through increases in evaporation and decreases in runoff that have potentially significant local hydrological effects on water availability down-stream. Case studies demonstrate that there are large variations in response to afforestation and reforestation (as a means to sequester carbon), with many positive feedbacks, thus providing potential trade-offs and synergies between environmental management and climate change mitigation. On the positive side, afforestation can result in changes to run-off and erosion (reducing flood risk), increased control of nutrient fluxes and increased water quality, but these could be offset by lower baseflows. Greater scientific awareness of the interconnectivities between forestry, hydrological and carbon systems and processes would be beneficial.

Further trade-offs between (re)afforestation and other sectors include the risk of forest fire (releasing stored carbon, removing a carbon store and changing the water regime), timing of harvesting and the cost of water, and the change and prioritisation of land use (such as the production of bio-fuel, agriculture, hydrology, GHG management and waste management; Ostle *et al.,* 2009).

10.4 Water

As was seen with the cross-sectoral interactions, water is fundamental to all the sectors. One way to adapt and increase freshwater resources is desalinisation, but this could lead to a number of trade-offs. For example, there is an increase in greenhouse gases, pollutants and changes to water prices and geopolitics (McEvoy and Wilder, 2012). It also has the potential to increase urbanisation, as water can be distributed to a larger population. McEvoy and Wilder (2012) also note that there is an increased reliance on technical expertise and reduced opportunity for decision-making, thus leading to reduced flexibility. Such management is contrary to the 'Dublin Principles' of a more participatory, decentralised water management strategy. A fair pricing scheme, education and water conservation measures, and alternative energy sources should be undertaken to not limit a region's adaptive capacity so that the poorest and most vulnerable members of the community can benefit. Finally, as far as flooding is concerned, setting aside land to store floodwaters (which could be designated under EU directives) can potentially change the biodiversity.

11. Evidence on the effects of timings of actions

The timing of implementation and the effectiveness of adaptation and mitigation measures is very much sector-specific, with some, such as changes in crop management, able to be implemented relatively quickly, i.e. within a year, and others, such as hard engineering options, requiring more planning and a longer lead time. All sectors, however, display a range of timings, depending on the measure under consideration and, thus, this section is analysed by sector.

11.1 Evidence by Sector

11.1.1 Agriculture

In agriculture, it has been suggested that adaptation can occur on various time scales (Table 11), with tactical actions being short-term, whereas strategic approaches that require the development of policies, institutions and infrastructure to allow regional adaptation of agriculture will occur over longer timescales (Howden *et al.*, 2007), as will changes in land use where the adaptation of existing uses is no longer viable (Gifford *et al.*, 1996).

Table 11: Speed of adoption of major adaptation measures (from Reilly, 1995).

Adaptation	**Adjustment time (years)**
Transportation system	3-5
Opening new lands	3-10
Variety adoption	3-14
Variety development	8-15
Fertilizer adoption	10
Tillage systems	10-12
New crop adoptions: soybeans	15-30
Irrigation equipment	20-25
Dams and irrigation	50-100

Changing sowing times is a very short-term strategy that can be initiated by individual farmers in response to changing climate trends (Trnka *et al.*, 2004), while switching between winter and spring sown crops may require different investments, e.g. in machinery and management practices; this might be considered as a medium-term option (Howden *et al.*

2007; Wolfe *et al.* 2008). Many of the breeding actions involve medium to long-term research and testing (Reilly, 1995), although once suitable cultivars or breeds have been developed they can be used in the short-term. A comparison of heat tolerance in adapted local and high-yielding breeds suggested that the latter are better prospects for climate change, as their heat tolerance can be improved in a few generations, whereas local breeds need more than 30 generations to reach a comparable milk production (Nardone and Valentini, 2000). For agricultural water use many of the actions are taken in response to changing weather conditions, but the change to using more water-efficient forage species in response to drought in Australia took place over three to five years (Henry *et al.*, 2012). The implementation of structural alterations (e.g. construction of retention reservoirs and dams) for water supply will take longer.

Howden *et al.* (2007) emphasise the need to align the spatial, temporal, and sectoral scales and the reliability of the information with the scale and nature of the decision. For example, for short-term adaptation actions by farmers, longer term climate projections may be of less use than short-term local weather forecasting. It is, however, important that short-term actions do not preclude longer term options.

11.1.2 Biodiversity

Modelled projections of the potential impacts of climate change on species have shown that the climatic range of 1,200 European plant species could contract by 6-11% over the next 50 years (Araujo *et al.*, 2004). While in Germany, it is predicted that as much as 30% of the country's current plant and animal species could become extinct in a timeframe of decades as a result of climate change, with those in the Wadden Sea tidal flats being particularly vulnerable (BMU, 2008). This means that adaptation measures are needed in order to try and avoid some of these impacts and, as has been discussed in Section 3.2, current measures may not be sufficient to enable species to adapt autonomously through dispersal (Vos *et al.*, 2008). While no particular evidence was found in this review of the timings of the adaptation measures, most of them, such as corridors and networks, are likely to take a number of years to fully implement *and* to become effective. More information, however, was found for coastal ecosystems, as discussed in the next section.

11.1.3 Coasts

The majority of schemes in this review focused on the transformation towards a more natural and dynamic coastline, with examples of numerous habitat creation and restoration schemes. Details on the specific timings for the design and implementation of schemes are rarely discussed in the literature, the only example being two years for the completion of a saltmarsh restoration project in the US (Hinkle and Mitsch, 2005). It also seems important to consider the timescales associated with recovery; both ecologically and in terms of reaching a dynamic equilibrium. This timescale will be largely dependent on the definition of recovery, with Barkowski *et al.* (2009) reporting the re-establishment of saltmarsh after a few years, whereas the recovery measures, such as species richness and resemblance to a natural marsh system, require a much longer timeframe. Management is another factor to consider with planned restoration schemes, especially those as part of compensatory habitat schemes being highly managed, having certain goals and targets, employing actions such as seed planting (Hardaway *et al.*, 2010; Nordstrom *et al.*, 2007), and the installation of fish ladders (Balletto *et al.*, 2005) to achieve these within a set timeframe (Hinkle and Mitsch, 2005; Wolters *et al.*, 2005a; Wolters *et al.*, 2008). Autonomous adaptations in contrast are likely to receive little or no management and therefore recovery will occur at slower, natural rates. One further factor to consider is the extent to which a scheme alters the environment, with managed retreat leading to a more natural regime and high impact schemes, such as the Delta Project, dramatically altering the coastal environment by enforcing a highly unnatural regime (Louters *et al.*, 1998; Smits *et al.*, 2006).

11.2 Ecological recovery

This review found different recovery rates for individual species, for example, the colonization by annual species (e.g. *Suaeda maritima*) is more rapid than that of perennial species (e.g. *Aster tripolium*), which require three years to establish (Wolters *et al.*, 2008). A study detailing a wetland restoration scheme in the US also showed a difference between species, with *Spartina* marsh recovering within 0.2-1.2 years, whereas mudflat vegetation required a longer period of 1.6-2.2 years (Janousek *et al.*, 2007).

Ecosystem recovery time was found to vary greatly between studies, ranging from only a few months (Darnell and Heilman, 2007) to a century (Garbutt and Wolters, 2008). Authors reporting rapid recovery times after managed realignment include Reading *et al.* (2008), with the establishment of fourteen intertidal invertebrates, after just two months; and Mazik *et al.* (2010), who report the total number of species to equal that of a reference marsh after one year. Pethick (2002) quotes a period of two years for saltmarsh vegetation to be fully established, whereas a study in the US found full establishment to require three years (van Proosdij *et al.*, 2010). A three year recovery period was also cited for the return of a functional waterbird assemblage on the Humber Estuary, UK, after realignment, although many habitats were still developing after this period (Mander *et al.*, 2007). One study suggests that recovery in terms of plant colonisation is usually achieved within five years (Reading *et al.*, 2008), with the same time reported for species diversity to be equivalent to a reference marsh after realignment (Wolters *et al.*, 2008). In contrast to these short recovery periods after realignment, the length of time required for vegetation recovery after the construction of the Oosterschelde Barrier, The Netherlands, was double that reported by Reading *et al.* (2008), and implies that ecosystems take longer to recover from high impact schemes (Noordwijk-Puijk *et al.*, 1979).

Studies quoting more extensive recovery periods include Hampel *et al.* (2003), with ten years for a new marsh system to develop after a natural dyke breach; and Warren *et al.* (2002) reporting fifteen years for avian breeding populations to be comparable with that of existing marsh. A number of studies agreed recovery was a long-term process, more suitably considered over timescales of decades (Hansen, 2009; Roman *et al.* 2002). Walker and Campbell (2010) found data to suggest that created marsh did not host the same fungal communities as natural marsh after 26 years, and Lee (2001) suggests a period of up to 50 years for habitat of an international standard to develop.

These examples highlight the broad range and variability of the recovery times of ecological systems, with some evidence of some species recovering in only a few months and others requiring significantly longer. There is, however, evidence to suggest that recovery is a long-term process, which should be assessed on a timescale of decades, and perhaps as a result, it is advisable that management of schemes impacting biodiversity give careful consideration to this.

11.3 Equilibrium development

11.3.1 Coasts

A study from the US, in which tidal channels were created as part of a saltmarsh restoration project reported all channels to have developed as intended within a period of 1-2 years (Teal and Weishar, 2005), whereas four years after managed realignment at Freiston Shore, Lincolnshire, UK, there was no evidence that a hydro-geomorphological equilibrium had been reached (Rotman *et al.*, 2008). At the Orplands Farm site in Essex, UK, managed realignment restored the tidal flow to an area used for agriculture, however, after an eight year period soils below 4-6 cm depth still retained many properties of agricultural soil (Spencer *et al.*, 2008). In contrast to these rather short time frames, van Dyke and Wasson (2005) predict it will take decades for the coastal system at one site in the US to reach equilibrium after levees were breached. The recovery timescale proposed after the construction of a storm-surge barrier in the Netherlands was the longest identified in this review, Louters *et al.* (1998) predicting that the establishment of a new hydraulic and geomorphologic equilibrium will span a period of centuries or longer.

Sediment accretion rates on restored sites are reported to be controlled by numerous factors including tidal elevation, water table height, natural accretion rates of the area, and slope (Warren *et al.*, 2002). Mazik *et al.* (2010) suggest that for a dynamic estuary such as the Humber, UK, with both high tidal elevation and accretion rates, managed realignment is only a temporary solution. In addition, the study reported that due to low sediment accretion less than half of the planned 45 ha of intertidal mudflats had developed six years after realignment. At the Freiston Shore managed realignment site, Symonds and Collins (2007a) observed a period of fourteen months before the tidal curve resembled that of natural saltmarsh, with channels beginning to achieve hydrodynamic equilibrium after twenty-seven months.

11.3.2 Forests

Adaptation options can be either short- or long-term. According to the FAO (2007), short-term measures can be considered as the various autonomous interventions, where no other sectors are involved, while long-term measures are characterised by structural changes such as forest fire management measures, the promotion of agro-forestry, and adaptive management with suitable species and plantations. Forest restoration is considered mainly as a long-term process because it can last for a period of a century or more. Therefore, it is essential to plan on long-term returns on restoration investments if the forests intend to support a wide range of species, species interactions and ecosystem services which are present in current forests.

The recovery rate of forests depends on the resources devoted to restoration, although not all restoration projects are undertaken for climate change mitigation/adaptation. Studies suggest that forests with a high number of species experience a faster recovery rate due to additional colonisation by species from the surrounding environment (Dale *et al.*, 2001). Forests with few species may face higher risks from fire and grazing (Dale *et al.*, 2001). Recovery can be highly influenced by location and, according to Lamb and Don Gilmour (2003), restoration can be difficult on sites with strongly seasonal climates or low soil fertility.

Some typical case studies are presented regarding the rate of rehabilitation of forests. In the Czech Republic, the restoration of forest species richness on mine spoils took 20-30 years to completion. Slower regeneration of seasonal deciduous forests was observed in Brazil utilizing techniques involving plowing and mechanical planting (Prach *et al*., 2007). Chazdon (2009) concluded that an aggressive global program of reforestation and natural regeneration could potentially restore forests on 700 million ha over the next 50 years. Fast growing, short-lived species with low-density wood are preferred by many reforestation projects designed to provide carbon offsets, but long-term carbon sequestration is promoted by growth of long-lived, slow-growing tree species, with dense wood and slow turnover of woody tissues.

11.3.3 Urban

There was very little evidence found for the urban sector on the timing of measures and this related only to green roofs. For example, the average lifetime of a green roof exceeds that of conventional roofing systems as the vegetation layer reduces the amount of UV reaching building materials, preventing their deterioration (Oberndorfer *et al.,* 2007; Ottelé *et al.*, 2011). As a result of these and other factors, over a period of 40 years a green roof is thought to be 25-40% cheaper than a conventional roofing system (Clark *et al.,* 2008). Also, the potential for carbon sequestration of green roofs is somewhat limited, as over time a green roof system will reach a carbon equilibrium, and no longer function as a carbon sink (Rowe, 2011).

11.3.4 Water

Timing of adaptation and mitigation measures is variable. The majority of studies regarding adaptation discuss schemes which have happened and their effects (e.g. Hansford, 2004; European Environment Agency, 2007; Howgate and Kenyon, 2009; Lehner *et al.*, 2011). Fewer studies investigated future long-term adaptation (e.g. Kirshen, 2007 who looked at a water resource adaptation for the 2030s, but with a planning scenario in the 2050s. See Section 15.1.6). For mitigation, the emphasis is on schemes – whether in action or in the planning stages and their possible effects. Frequently this is over long time scales – up to a century - as this is where the greatest benefits are seen (e.g. Fischer *et al.*, 2007), rather than the near-term benefits which are assessed for adaptation (e.g. Dessai and Hulme, 2007).

Charlton and Arnell (2011) state that climate change is the largest single driver of future water supply in England, and thus over a planning period of supply water, climate must be taken into account. Not all countries are the same, and water abstraction and storage could be the largest changes to a system. For instance, groundwater withdrawal has led to subsidence threatening infrastructure (particularly in Asian cities). In Bangkok, Thailand, water abstraction was 8630 m^3 day^{-1} in 1954 and over the following decades increased to over 1 million 8630 m^3 day^{-1}. This resulted in subsidence, with local rates up to 35 mm y^{-1} (das Gupta and Babel, 2005). City planners realised this was not sustainable, so reduced water abstraction and the building of new wells. Lehner *et al.* (2011) report that 6,800 reservoirs have been built world-wide which can help regulate water supply, but these also affect local or regional hydrological regimes. Ideally, adaptation (and mitigation) schemes need to be assessed over the lifetime of operation, remembering to consider the wider implications of a scheme, potentially including the whole floodplain or river basin. For ecosystems this could take many decades, or for some forms of adaptation (e.g. building a large dam that fundamentally changes the drainage system) ecosystems never fully recover from human influence.

12. Evidence on scale at which action is taken

According to Adger *et al.* (2005) adaptation to climate change can be implemented by various agents, from individuals, firms and civil society, to public bodies and governments at local, regional and national scales, and international agencies. This section, therefore, examines the scale of measures and their implementation sector by sector in order to draw out some of these differences.

12.1 Agriculture

For agriculture, the scale at which adaptation is undertaken varies from the field (Gaydon *et al.*, 2012), through farm (Bryan et al, 2009; Fleisher *et al.*, 2011), to basin level (Quiggin *et al.*, 2010), often depending on the actors involved (see below). Implementation of many of the various adaptation measures is at the farm-scale, but the development of new cultivars is likely to be at the (inter)national level. The development and exploitation of genetic diversity for adaptation to climate change also occurs at several scales. As noted earlier, this development is fundamentally dependent on an individual species genetic diversity. There are a number of global initiatives, such as gene banks, aimed at conserving this diversity. For example, ICARDA (International Center for Agricultural Research in the Dry Areas) has more than 120,000 accessions of species, including important food and feed crops, such as barley, wheat, lentil, chickpea and vetch. Often in parallel are efforts to maintain diversity *in situ,* as it has been found that landraces and, when available, crop wild relatives harbour a large amount of genetic variation, some of which can be used immediately in breeding for drought and high temperature resistance (Ceccarelli *et al.*, 1991). There are several international projects aiming to identify the genes associated with superior adaptation to higher temperatures and drought to aid adaptation (Ortiz, 2008; Ceccarelli *et al*. 2010). These two methods of conserving genetic variation differ in that the gene banks contain the genetic information present at time of collection, while maintaining diversity *in situ* is dynamic, as species continue to interact with their environment and can generate continuously novel genetic variation (Ceccarelli *et al.*, 2010).

Alongside international efforts is more regional and national research aimed at identifying sources of genetic diversity for application at a more local level, with testing being done at the field scale. In Romania, for example, the National Agricultural Research and Development Institute (NARDI), Fundulea (Verzea, 2007), maintains large working collections of local populations and cultivars of crops, such as wheat maize, sunflower and forage crops, seeking to improve them. Genetic progress for wheat yield is estimated at about 50 kg ha^{-1} y^{-1} or 1% y^{-1} and average maize yield increased from 1,270 kg ha^{-1} (1951-1955) to 1,770 kg ha^{-1} (1961-1965) and was maintained at a high level between 1978-1996, with increases of 83 kg ha^{-1} y^{-1}. It also exchanges germplasm with international breeding centres (e.g. CIMMYT, ICARDA).

12.2 Biodiversity

In response to international commitments, such as the Convention on Biological Diversity and the EU Birds and Habitats Directives, there are a number of measures aimed at conserving biodiversity, which, as has been discussed, also are relevant to helping it adapt to climate change (Section 3.2). At the European scale, the Natura 2000 network[7] is one programme seeking to improve connectivity across Europe. The European Green Belt (EGB) is another biodiversity network acting on a European scale. This network operates across 24

[7] http://www.natura.org/about.html Accessed 06/08/2012

countries, aiming to increase cross-border connectivity, thus aiding the dispersal of species across areas of Europe (Zmelik *et al.*, 2011). Networks and corridors can also be implemented at the national scale, e.g. in the Netherlands (WRD, 2011). Also, as part of the German Adaptation to Climate Change Strategy and the National Strategy on Biological Diversity, the German Federal Government recognises that Länder should improve networks to allow species and populations to migrate northwards in response to climate change, and are therefore taking precautionary measures to aid adaptation (BMU, 2008). Other adaptation actions, such as assisted migration may occur primarily at a more regional level, and habitat restoration at a more local level.

12.3 Coasts

The majority of adaptation interventions covered in the literature, particularly those in Europe were implemented at a local scale (Bulleri and Chapman, 2004; da Silva and Duck, 2001; Klein *et al.*, 1998; Lamberti *et al.*, 2005). Restoration projects generally covered an area of up to a few hundred hectares and include 21 ha saltmarsh creation at Bidasoa, Spain (Marquiegui and Aguirrezabalaga, 2009), the restoration of 30 ha of intertidal marsh on the Scheldt Estuary, Belgium (Verbessem *et al.*, 2007), and 400 ha of tidal marsh reclamation at Blackwater, UK (Townend and Pethick, 2002). Breakwater schemes were also relatively small in scale, with examples of structures protecting 1.1 km, 4.5 km, and 5.9 km of coastline (Lamberti *et al.*, 2005; Saiz-Salinas and Urkiaga-Alberdi, 1999; Vandenbroeck, 2006).

The managed realignment of coastal defences is conducted on an estuary-wide scale, with individual plans for specific locations along the estuary (Reading *et al.*, 2008; Spencer *et al.*, 2008; Wolters *et al.*, 2008). The largest scheme discussed in the UK was the realignment of over 100 km of defences along the Humber Estuary, with the subsequent creation of 7,494 ha of (i.e. a 69% increase in) intertidal habitat (Andrews *et al.*, 2008). The literature shows that the scale of projects undertaken in the Wadden Sea area are much larger than this, with a number of regional plans including de-embankment strategies (Barkowski *et al.*, 2009; Wolters *et al.*, 2005a), and saltmarsh creation covering 450 km of coastline in Germany (Hofstede, 2003). The largest coastal adaptation measure identified is the Wadden Sea Plan - the only example found in this review of trilateral coastal management, with Germany, Denmark and the Netherlands legislating common policy and creating joint targets for the protection and conservation of the Wadden Sea area (Enemark, 2005; Falk *et al.*, 1994).

At a national level, a promotional intertidal agri-environmental scheme led by the UK government was created to meet national targets and reduce the loss of valuable saltmarsh habitat (Parrott and Burningham, 2008). This scheme recognized the value of coastal saltmarsh, giving landowners the opportunity of receiving payment from DEFRA by giving their land for habitat creation as part of a managed realignment scheme.

From a global perspective, schemes in the US appear to be taking place on a national scale, with Hansen (2009) describing a scheme with two million acres of land earmarked for wetland restoration through the USDA’s (United States Department of Agriculture) Wetland Reserve Program. Further examples include the Coastal Program and National Coastal Wetlands Conservation Grant Programs, both governed by the US Fish and Wildlife Service, and having restored over 190,000 acres of coastal wetland habitat since 1990 (Mangin and Valdes, 2005). In contrast to these examples of national scale adaptation, the review did identify one instance of actions taken at the individual level, with some land owners in New Hampshire, US, constructing walls to protect their properties from the sea (Bozek and Burdick, 2005).

12.4 Forests

In the case of the forest sector, the majority of adaptation measures are implemented at regional and local levels and a selection of the case studies identified in this review are given in Table 12. The classification of projects in forestry, according to their scale, depends on the geographic coverage of project implementation, i.e. regional, national or local, and the scale can refer to specific ecosystems or to political-administrative divisions. Finally, Chazdon (2009) stated that forest restoration efforts, whether at national, regional, or local scales, will take many decades, long-term financing, political will, labour, and personal commitment.

Robledo and Forner (2005) specified three types of projects regarding adaptation stages consisting of vulnerability assessments of ecosystems and the forest sector, improvement in the capacity and design of measures, and implementation of adaptation measures for improving the adaptive capacity of ecosystems and the forest sector.

Table 12: Forest adaptation projects classified according to scale.

Scale	Project
National	National training plan on adaptation for the forest sector
National	Large scale reforestation with poplars in small holder woodlots, agroforestry productions systems and watersheds in China
Regional	Assessment of vulnerability to climate change in the European Union
Regional	UNDP-GEF project for the improvement of training on adaptation (Central America, Mexico and Cuba)
Local	Vulnerability assessment of a micro watershed
Local	Study of the impact of climate change on the marketing of agricultural products in two departments
Local	Project to pay for soil conservation in a micro watershed

12.5 Urban

In the urban sector it was found that many schemes are financially supported or policy driven by local (e.g. Beddington Zero Energy Development, UK (Section 4.5.9), regional or national (e.g. SUDS-based retrofitting project in the Augustenborg neighbourhood, Sweden (Section 3.5.7); new build on brownfield sites, UK, (Section 4.5.3)) governments. Most of the urban adaptation and mitigation measures, such as rainwater harvesting and greywater usage, were more applicable and effective at the household to neighbourhood, rather than regional scale (Farreny *et al.*, 2011), but they both help in decentralising the water supply, reducing potable water use (Wise *et al.,* 2010) and increasing regional resilience to drought by improving water security (Graddon *et al.,* 2011). There are numerous examples of solar energy being used in housing developments of various scales and design throughout Europe, e.g. PV panels can be installed by individual householders or they may be part of a wider community-based development. If, however, low energy residential settlements, such as the Beddington Zero Energy Development, UK, were implemented on a European-wide scale, it is estimated that reductions of 90% of CO_2 emissions could be achieved without adverse impacts on the quality of life of residents (Chance, 2009).

12.6 Water

When considering water resources, climate change (and associated adaptation and mitigation) needs to be considered alongside changes to land cover and water demand. Estimates that have been made range from global scale (e.g. Fischer *et al.*, 2007), regional scale (e.g. Kirshen, 2007), country level (e.g. Charlton and Arnell, 2011), basin level (e.g. Howgate and Kenyon, 2009), and settlement or local level (e.g. Cooper and Knight, 1990; Stalenberg, 2012). Thus Todd *et al.* (2011) concluded that water resource adaptation studies should happen at many different scales. A similar situation occurs with respect to mitigation. However, whilst adaptation can often be viewed as a local issue, it is best viewed in the context of other drivers, as adaptation decisions can have secondary impacts affecting a wider area. Subsequently, a 'source-pathway-receptor-consequence' model is used to view climate and socio-economic risk in the context of other drivers (Evans *et al.*, 2004).

Charlton and Arnell (2011) report that there are very few studies that look at how water-orientated organisations are adapting to climate change and how they are achieving this. Many developed countries have good investment in their water resources and monitor them, so consider their long-term utilisation. This can include local, national or international policies (e.g. the EU Habitats Directive and the Water Framework Directive). Over the long-term, water security infrastructure in developed countries may need re-engineering to protect biodiversity (Vörösmarty *et al.*, 2010). Conversely, reduced investment in developing countries, the remoteness of some localities and relative cost of water, make them vulnerable to changes in water resources as they are less resilient to environmental change (Vörösmarty *et al.*, 2010; Joint Research Centre, 2012). Investment in water infrastructure must also protect biodiversity, creating a dual challenge for engineers and scientists (Vörösmarty *et al.*, 2010). International aid and increasing scientific awareness is providing improved support for assessing future water availability, the likelihood of floods and drought, and water scarcity (Joint Research Centre, 2012).

This review found that the scale at which adaptation is undertaken varies from the local to international according to the sector and the measure being considered. The scale also interacts with the actors involved who can range from individuals through communities, to national and international bodies. They can drive adaptation or mitigation or can respond to legislation, policy or financial incentives for such actions.

13. Actors involved

Little explicit discussion was found on the actors involved in adaptation and mitigation. In agriculture, it has been suggested that adaptation involves several different actors and many organisational levels from international institutions, governments, agri-business to individual farmers and that each level of actor has a different role to play (Gifford *et al.*, 1996; Smit and Skinner, 2002). The prime actors identified through the review were farmers, as many measures are implemented at the field and farm scales (e.g. Southworth *et al.*, 2002; Howden *et al.*, 2007; Wolfe *et al.*, 2008; Fleisher *et al.*, 2011; Gaydon *et al.*, 2012), while those at a larger scale can involve water boards or governments encouraging or subsidising particular actions (Mushtaq and Moghaddasi, 2011). Some of the longer-term adaptations, including the development of new cultivars, forecasting and advice on management, will involve researchers, meteorological services and government advice (Trnka *et al.*, 2004). Smit and Skinner (2002) in a review of the Canadian situation suggest that most cultivar development has been done by the private sector.

Some information on the actors involved in adaptation and mitigation could be gathered from examining who was involved in implementing particular measures. In the coastal sector, several adaptation schemes involved partnership working across a number of organisations. For example, the Hesketh Out Marsh West, UK, managed realignment scheme involved the Environment Agency (a UK Government Agency), RSPB (a NGO) and Morecambe County Council (Tovey *et al.,* 2009), while a similar scheme on the River Humber at Welwick involved the Environment Agency, Natural England (an Executive Non-departmental Public Body) and RSPB (Pontee, 2007). Some of the measures in urban areas, such as rainwater harvesting and solar panels may be undertaken by individual householders, developers or local councils.

14. Governance

According to Adger *et al.* (2005), adaptation to climate change can be implemented by various agents from individuals, firms and civil society, to public bodies and governments at local, regional and national scales, and international agencies. In the papers reviewed, little mention was made of the governance level and, thus, only a snapshot can be presented based on the articles reviewed.

14.1 Agriculture

Smit and Skinner (2002) suggest that adaptation in agriculture is often seen as a government policy response, but that decision-making also involves agri-business and producers at the farm-level. Gifford *et al.* (1996) suggest that Governments should consider long-term scenarios and their implications, and policies that influence bank lending arrangements. To this could be added land use planning and policies that affect a number of sectors, and there has been a call for mainstreaming adaptation within and across sectors (e.g. Howden, 2007). While individual farmers cannot plan long-term for highly uncertain specific scenarios, they can respond to short-term external events; influencing them and build flexibility into their management. At the level of agricultural research, potential proactive and reactive adaptation options need to be investigated and developed.

Agriculturalists have to work within EU and national legislative frameworks, and the same of course would be true for actors in the other sectors. For example, breeders in organic agriculture have to work within the framework of organic farming (Council Regulation (EC), 2007, No 834/2007), which can be difficult to adapt to the local situation (Wolfe *et al.*, 2008). Also, while most organic farmers depend on modern varieties bred for conventional agricultural systems, the European Organic Seed Regulation (EC 1452/2003) is making the use of organic seeds compulsory.

14.2 Coasts

In the coastal sector, the governance was focused heavily at the national level, due to coastal planning occurring at this level even though many of the schemes were local and involved other actors. In the Netherlands, schemes such as the Wadden Sea (Enemark, 2005) and the construction of dams, sluices and storm-surge barriers (Elgershuizen, 1981; Wolff, 1992) are undertaken by the national government. The same is true in the UK, with schemes such as managed realignment at Orplands (Emmerson *et al.*, 1997) and Wallasea Island (Dixon *et al.*, 2008), and this is also seen in the US schemes (Hansen, 2009). While near Rome, Italy, beach nourishment and an offshore underwater rock barrier were undertaken by the Office for Civil Engineers of Maritime works, Rome and the Italian Ministry of Public Works (Lamberti *et al.*, 2005).

14.3 Urban

In urban areas, local regulations have been used in the case of rainwater harvesting schemes (RWHS), with the Sant Cugat del Vallès municipality being the first in Spain to change the building code through local regulations, mandating all buildings with over 300 m^2 garden to install a RWHS (Domènech *et al.*, 2011). In addition, since 2002, all newly built dwellings with over eight apartments, or an annual shower water consumption of over 400 m^3 are required to install a rooftop RWHS to re-use the greywater from the shower for toilet flushing (Domènech and Sauri, 2011). Research has shown that a water tank of 70 m^3 volume would be sufficient to irrigate a communal garden of 300 m^2 (Domènech and Sauri, 2011). In addition, a tank of 6 m^3 in a single family house would be able to supply 100% of the laundry water requirements, with water savings of 16 litres per capita per day (Domènech and Sauri, 2011), but this is expensive (Section 15.1.5). After the success of Sant Cugat del Vallès RWHS, the uptake of water recycling systems in Spain has increased, with over 40 municipalities in the region of Catalonia enforcing local regulations to encourage the installation of these systems in new buildings (Domènech and Sauri, 2011). Much urban planning, however, has an important national dimension too.

15. Other impacts of adaptation and mitigation actions

Adaptation and mitigation actions, in addition to their impacts on climate change, also carry with them a range of economic (see Section 15.1), environmental (Section 15.2) and social (Section 15.3) effects.

15.1 The economics of mitigation and adaptation

This review found some evidence of the costs of adaptation (and mitigation), which varied according to the measure concerned. For this reason, costings will be examined by sector, before seeking some commonalities.

15.1.1 Agriculture

Wolfe *et al.* (2008) have identified a number of economic consequences of adaptation including:

- changing the time of planting could be economically disadvantageous to the farmer if it results in taking the harvest to market when prices are lower due to the supply/demand balance;
- changing to perennial crops is a more expensive option, as new plants will take several years to reach their maximum productivity;
- new stress-tolerant seeds may be expensive to purchase and may require investment in new equipment or changes in farm practices.

Changing the sowing date had also been seen as a no-cost option, although if the change is too large then it could impact on the management of other crops (Alexandrov *et al.*, 2002). In the case of water management it has been suggested that while irrigation is a viable adaptation measure, it may not always be economically viable. This was found to be the case by Finger *et al.* (2011) exploring the future of Swiss maize farming, and the adoption of irrigation technology was dependent on the level of Government support, while the economic benefits of irrigation were very sensitive to crop and water prices. Others suggest that while there are many structural adaptations for increasing water storage, it is difficult to quantify the associated costs and they call for cost-benefit analyses (Moriondo *et al.*, 2010). Also, long cycle cultivars can demand 25-40% more water, which may not be available or be cost effective in the future (Giannakopoulos, 2009). In contrast to these negative consequences, the planting of shade trees for livestock can increase farm income through sale of wood products and potential tourism due to landscape improvement (Iglesias *et al.*, 2007).

As far as farm economics are concerned, no-till provides an opportunity to reduce costs from fuel and machinery, as a result of minimal soil disturbance and lower production costs (Antle *et al.*, 2012; Bescansa *et al.*, 2006; Desjardins *et al.*, 2005; Soane *et al.*, 2012). Furthermore, a study examining the potential for reduced tillage in north-east Germany has concluded that the extended use of this management technique would improve the profitability of crop production (Verch *et al.*, 2009). However, it is interesting to note, that from the perspective of climate change mitigation, it is thought that neither a market-based emissions trading scheme, nor government subsidies would be able to accurately reflect the abatement of GHG emissions, with mitigation being underestimated in most wheat cropping systems and overstated in many corn-soy-hay systems (Antle *et al.*, 2012).

In China, a study in the Yangtze River basin observed that to cope with increasing water stress, farmers chose the most cost-effective options from a range of possible responses based on their intuitive calculation of relative cost and expected returns (Liu *et al.*, 2008). Liu *et al.* (2008) even made a cost-benefit analysis of the major coping responses, from which it can be seen that most adaptation measures have positive economic benefits (Table 13). Similar results come from an economic evaluation of a GEF (Global Environmental Facility) project in North Jiangsu, which concluded that the adaptive activities, including rainfall collection projects, soil water saving technology, breeding selection, climate change training, etc., will contribute increases of 89.5 kg ha^{-1} and 636.2 kg ha^{-1} in local wheat and rice yields respectively (Zhang *et al.*, 2011).

Table 13: Cost-benefit analysis of major adaptive response in Lower Yellow River (Liu *et al.*, 2008).

Response	Cost (RMB/Mu*)	Benefit (RMB/Mu*)
Mulching (plastic sheet)	20-40	50-80
Improved drought-resistant variety	50	70-100
Increased fertilizer investment	50-100	50-100
Increased cotton sown area	200	200-300
Growing winter dates	1,500-2,000	2,500-3,500
Raising pigs	400-500	1,500-2,000
Growing vegetables in greenhouses	5,000-6,000	10,000-12,000

** 1MU=1/15 ha*

Conservation tillage can reduce the labour input and, thus, reduce the cost of crop production. Some field experiments show that it can reduce the average cost of crop production by 600-1200 RMB ha^{-1} for rice (Gao, 2011), 375-450 RMB ha^{-1} for maize and soybean (Wang *et al.*, 2010b), and 300-450 RMB ha^{-1} for wheat (Wang *et al.*, 2010b). The net income from crops as a result of conservation tillage could increase by 0.95-12.42% in southwest China (Zou *et al.*, 2010) and 49% in northern China (He *et al.*, 2006).

As far as an instrument to encourage the uptake of mitigation practices, Neufeldt and Schäfer (2008) used a regional economic-ecosystem model to assess the impact of various mitigation strategies, such an emissions tax, for the agricultural sector in south-west Germany. The authors found that for Baden-Württemberg, the introduction of a nitrogen-tax would have the largest impact on fertiliser production based emissions, but relatively little effect on farm systems associated with animal husbandry. It was also found that in farms growing forage, this strategy was less effective as these utilised manure and had fewer animal feed crops. The study found in contrast, that both CO_2 and CH_4 emissions from livestock declined when a general emissions tax was introduced, as the animals were given less additional feed. This, therefore, saw a larger amount of abatement from animal husbandry than associated with a specific tax on nitrogen (Neufeldt and Schäfer, 2008). Finally, the model showed that an overall emissions cap would result in a 12% GHG mitigation for the region by reducing the use of synthetic nitrogen fertilisers as well as livestock numbers (Neufeldt and Schäfer, 2008). In a similar study, Durandeau *et al.* (2010) found that introducing a tax on livestock, or nitrogen fertiliser consumption was the second best policy for reducing GHG emissions, with a higher abatement potential when the tax was greater than €200 per livestock unit, or per ton of nitrogen fertiliser.

15.1.2 Biodiversity

Few costings were found for the adaptation measures considered for biodiversity, but as many of the measures are part of good conservation practice it is difficult to assess what is explicitly undertaken for climate (Berry, 2009b). The de Doorbraak project (Box 1, Section 8.2.1) was commissioned by the Regge en Dubjek district water boards, the province of Overijssel, and the Ministry of Agriculture, Nature Management and Fisheries, at a cost of €40 million (WRD, 2011). While the scale of habitat restoration in the Restoring Peatlands Project (www.restoringpeatlands.org) varies between the two countries, the 14,000 ha peatland being rewetted in Belarus was financed at a total of €2.5 million by the Federal Republic of Germany as part of the International Climate Protection Initiative of the German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU); and 20,000 ha in the Ukraine at a cost of €4.9 million.

15.1.3 Coasts

The economic feasibility of adaptation and mitigation interventions is an important factor to consider in any management scheme. Projects leading towards a more natural coastline are seen in many cases as the most appropriate financial solution. This is because traditional adaptation measures such as sea-walls and beach nourishment are expensive, requiring both regular maintenance and management; seawalls for example costing approximately £400 per metre (Möller *et al.*, 2001); and construction costs for one LCS and beach nourishment scheme totalling €1 million (Lamberti *et al.*, 2005). Other adaptation options, such as storm-surge barriers carry with them considerable cost, with Smits *et al.* (2006) quoting the annual maintenance costs of the Oosterschelde storm-surge barrier to total €15 million, making it one of the most expensive engineering works to be completed in the Netherlands.

As a result of these high costs, hard-engineering schemes, unless protecting an area of extremely high economic value, are often seen as economically infeasible. This is reflected in the literature, with numerous examples of managed realignment being undertaken as the maintenance of existing defences was uneconomical (Emmerson *et al.*, 2000; Hazelden and Boorman, 2001; Klein and Bateman, 1998; Shepherd *et al.*, 2007).

The creation of saltmarsh and other wetland habitat through realignment schemes also has other financial incentives, through providing a coastal defence, and in some cases protecting existing structures (Hazelden and Boorman, 2001; Hofstede, 2003). King and Lester (1995) assessed this economic value of saltmarsh, calculating that if all saltmarsh in Essex were to disappear, a total of £600 million would be required for repairs to seawalls along the coastline (King and Lester, 1995). A linear negative relationship has been identified between saltmarsh width and seawall height, with a study by Colclough *et al.* (2003) calculating that for a marsh width of 80 m, a seawall height of 3 m, costing £400 per metre was necessary, whereas for a marsh width of only 6 m, a seawall height of 6 m at £1,500 per metre would be required. Ledoux (2003) compiled data to highlight the range of estimations for the defence value of coastal wetlands in monetary terms, with the majority of studies valuing this service at less than £400 ha^{-1} y^{-1}, which is significantly lower than the £7334 ha^{-1} y^{-1} reported by King and Lester (1995). Another aspect covered in the literature is the economic impact of current degradation and destruction of wetlands. For example, Lee (2001) estimated the cost of replacing the freshwater and brackish habitats in England and Wales lost as a result of coastal squeeze to be substantial, in the range of £50-60 million. In another study, a promotional intertidal agri-environmental scheme led by the UK government was created to meet national targets and reduce the loss of valuable saltmarsh habitat (Parrott and Burningham, 2008). This scheme recognized the value of coastal saltmarsh, giving landowners the opportunity of receiving payment from DEFRA by giving their land for habitat creation as part of a managed realignment scheme. The study showed that for saltmarsh creation over a ten year period, land owners could expect to earn £525 ha^{-1} y^{-1} for cultivated land, and £250 ha^{-1} y^{-1} for grassland (Parrot and Burningham, 2008).

Despite the global potential for wetlands to mitigate the effects of climate change, this area remains relatively poorly researched, with few studies identifying the actual economic value of these systems either in terms of carbon credits or cost of emissions avoidance. Luisetti *et al.* (2010) is one example of a study which gives a tangible value to the service of carbon storage in terms of the damage cost avoided per ton CO_2. Hansen (2009) also recognise the potential economic value of wetlands in carbon offsetting schemes, with an estimated value of US\$182-1,900 per acre for carbon offsets on a coastal flat in the US. Similarly, DeLaune and White (2011) calculated that the carbon storage benefits of 988,888 ha coastal wetland in Los Angeles, could amount annually to US\$29.7-44.5 million, whereas current national losses resulting from wetland deterioration and coastal squeeze could be costing the US US\$18.6-27.9 million per year. As far as the UK is concerned, Jickells *et al.* (2003) highlight the importance of wetlands on the Humber estuary, with land reclamation over the past 300 years resulting in considerable losses of sequestered carbon and associated loss of revenue from the sale of carbon credits.

Shepherd *et al.* (2007) used cost-benefit analysis to evaluate managed realignment on the Blackwater Estuary, UK. The authors totalled the costs of realignment at £811,893 per km, with additional economic benefits from a range of ecosystem services including habitat creation, carbon sequestration, the burial of contaminants, and a reduction in maintenance costs associated with hard-defences. Similarly, Dixon *et al.* (1998) reported the planning and construction costs of a seawall breach at Abbotts Hall as part of coastal realignment to total £75,000, part of which would be compensated for by large defence savings at other locations. From this aspect, saltmarsh has a high value, reported by King and Lester (1995) to be close to £6000 ha^{-1}, and exceeding the value of some grade I agricultural land. Klein and Bateman (1998) highlight the fact that although managed retreat schemes are viewed as being the most economically viable adaptation, it is inappropriate to assume that all schemes are economically efficient, with Shepherd *et al.* (2007) supporting this view by noting that this particular adaptation is only cost-effective if considered over the long-term.

As far as coastal mitigation is concerned, US studies highlight that the economic feasibility of wetland creation for carbon storage is dependent on market offset prices, competition for land and the costs of restoration, which can be substantial in a heavily managed restoration project (Hansen, 2009; Irving *et al.*, 2011; Yu and Chmura, 2009). It was calculated that 20-35% of forested wetlands in the Mississippi and coastal flats on the Gulf-Atlantic coast have carbon offset values greater than the cost of restoration (Hansen, 2009).

The use of wetlands for grazing can be combined with mitigation schemes to make them a more attractive financial option. This is possible as the effects of grazing are mostly limited to the above-ground biomass, whereas the majority of carbon is stored in the marsh soils (Connor *et al.*, 2001; Irving *et al.*, 2011; Olsen and Dausse, 2011). Yu and Chmura (2009) found that the introduction of grazing on a high-latitude marsh on the St Lawrence Estuary resulted in higher soil carbon density and below-ground productivity than in non-grazed marsh, making it a more effective carbon sink. This is due to grazing reducing the volume of above-ground biomass, subsequently resulting in increased soil temperatures, amount of evapotranspiration and better light conditions, all of which increase the below-ground carbon storage capacity of the soil (Olsen and Dausse, 2011; Yu and Chmura 2009). In contrast, a study concerning the impacts of wetland grazing in Denmark found that although the amount of net ecosystem production did not differ greatly between grazed and non-grazed sites, grazing significantly reduced the amount of organic matter stored in below-ground biomass, and as a result concluded that grazed marsh soils have a lower carbon storage capacity (Morris and Jensen, 1998).

Wetland creation or restoration, whether for the purpose of adaptation or mitigation, can be costly; however, the range of ecosystem services these coastal systems provide is also extensive. As far as wetland creation for mitigation is concerned, several studies suggest that funding the large-scale conservation of existing wetlands, with substantial existing stocks of underground carbon, may be a more appropriate action than costly small-scale restoration projects which take years to accumulate the same levels of carbon (Dyke and Wasson, 2005; Lee, 2001). Irving *et al.* (2011) support this view and identify the need for a substantial increase in the size of restoration projects if the global benefits of carbon storage from wetland restoration are to be seen.

From an economic perspective, a study from the US shows that restoration can benefit the fishery industry (Luisetti *et al.*, 2010), with one example after restoration at Galveston Bay in the US resulting in high densities of important fisheries species such as brown shrimp, white shrimp and blue crab (Rozas *et al.*, 2005).

15.1.4 Forests

Little information on the economics of the measures was found in the papers reviewed, although there was one case of the planting and protection of mangrove forests in Vietnam. Robledo *et al.* (2005) found that by 2005 nearly 12,000 hectares of mangroves had been planted, at a cost of $1.1 million, saving $7.3 million annually in dyke maintenance costs.

15.1.5 Urban

There are many examples in the urban sector of the cost-effectiveness (or otherwise) of adaptation and mitigation measures. Improvements in energy efficiency (Sections 4.5.4 - 4.5.6) can generate economic savings, as can energy savings associated with street trees (Section 15.1.5). Furthermore, the average lifetime of a green roof exceeds that of conventional roofing systems as the vegetation layer reduces the amount of UV reaching building materials, preventing their deterioration (Oberndorfer *et al.*, 2007; Ottelé *et al.*, 2011). As a result of these and other factors, over a period of 40 years a green roof is thought to be 25-40% cheaper than a conventional roofing system (Clark *et al.*, 2008). Other improvements to roofing, such as insulation can result in large savings associated with cooling and heating, and make this attractive from an economic perspective. For example, the potential savings from a study in Cyprus over a life-cycle, total up to €22,374 (Florides *et al.*, 2000).

Street trees in European cities have an average density of around 50-80 street trees per 1,000 inhabitants (Pauleit *et al.*, 2002). Although helping with urban adaptation to climate change, street trees can be associated with a number of adverse effects, including planting and maintenance costs (McPherson and Rowntree, 1993; Hegedüs et al. 2010; Tallis *et al.*, 2011). In Europe, research shows that the average costs to establish a tree (cost of tree and planting) vary considerably, with costs below €200 (e.g. Spain and UK) ranging to over €1,000 per tree (e.g. Norway, and Denmark) (Pauleit *et al*., 2002). Planted species also vary in their suitability to function in the urban area (Hegedüs *et al.*, 2011; Merse, 2009). Hence much better guidance needs to be given to tree planers in terms of tree selection and establishment; lowering cost and increasing suitability (Pauleit *et al*., 2002). A number of US studies consider street trees in economic terms, and despite the above management concerns, these remain on the whole an attractive option in economic terms. One example from Bismark (US) reporting a benefit-to-cost ratio of over 3:1, suggesting US$3.09 in benefits for every US$1 invested in management (McPherson *et al.*, 2005), while the the PG&E shade tree program had a high benefit-to-cost ratio, estimated over a 30-year period to be 19:3 (McPherson and Rowntree, 1993). To further make tree planting a more attractive economic option for city planners, Rowe (2011) suggests that green roofs could be incorporated as carbon trading credits under a cap and trade system.

Concrete slab cooling utilises heating ventilation and air-conditioning technology, allowing the thermal load accumulated during the day to be released at night via air coolers, with low energy consumption, maintenance costs and operational savings (Zimmerman and Anderson, 1998).

Low energy residential estates, such as BedZED (UK) and that discussed by Wojdyga (2009) for Warsaw (Poland), are associated with higher construction costs than standard buildings – estimated in the range of 7-10% for the latter development (Chance, 2009; Wojdyga, 2009). Despite this, such buildings are of high quality, using materials which have long life-times (Chance, 2009). Additionally, the low energy demand stemming from the design of such projects will prove economically beneficial for residents, with energy demand in the Polish project for consecutive heating seasons 2003-2007 being very low, totalling 31.9 kWh m^{-2} (Wojdyga, 2009).

Solar energy systems also have the potential to be cost-effective. The installation of solar PV and solar thermal systems on a hostel roof in Milan was simulated to meet a substantial proportion of the electricity demand for lighting and appliances (Adhikari *et al.*, 2011). The cost of installing two solar thermal collectors for hot water heating on the roof garden was around €3,533, with a 7-year payback period.

Despite these large potential GHG savings, solar PV systems are one of the most costly options for renewable energies, with a long payback period compared to, for example, wind energy, with 13 years payback for UK households compared to 2.8 years from wind energy (Allen and Hammond, 2010), and 81 years for domestic sites in the Eastside area at a cost of £305 million (Jefferson *et al.*, 2006).

Re-developing brownfield sites in existing urban areas also comes at considerable economic cost, being much more expensive than expanding onto a greenfield site. Additionally, as land becomes more scarce in the urban area, land prices are likely to increase (Birrell *et al.*, 2005; Searle, 2010), which could result in low-income residents being priced out of the city area (Ancell and Thompson, 2008).

While Rain Water Harvesting Schemes (RWHS) offer a number of possibilities for reducing the impacts of drought on water demand, especially in urban areas, they can be expensive. The initial cost of RWHS is high at €1,500-€4,000 for a 1,500-10,000 L storage tank (prices quoted for 2009), however, the cost of operation and maintenance is generally low (Li *et al.*, 2010). A 20 m^3 RWHS for landscape irrigation was estimated to cost €8,864 for a single family household or €633 per household in a multiple family building (Domènech and Sauri, 2011). As a result, and to make these systems more attractive to customers, the local authority in Barcelona offers subsidies of up to €1,200 to households which install RWHS, reducing the long payback period of these systems (Domènech *et al.*, 2011).

Greywater systems are again initially relatively expensive, with capital and installation costs ranging from €2,700-€3,400, and additional operational costs for chemicals, pumping and maintenance (Li *et al.*, 2010). Both systems have a long payback period, that in Ireland for RWHS ranging from 7-20 years (Li *et al.*, 2010), and 27 years estimated at a neighbourhood level for Granolles, Spain (Farreny *et al.*, 2011). As the cost of water may increase with future consumption and scarcity, RWHS appear to be an attractive option, for example, in countries such as Ireland where domestic water bills are to be reintroduced (Li *et al.*, 2010), and also in Spanish neighbourhoods where the adoption of RWHS could see annual water bills by the local authorities reduced (Farreny *et al.*, 2011).

As far as the economics of SUDS options for managing water runoff are concerned, in Glasgow a cost-benefit analysis revealed that the initial investment costs for these SUDS solutions was comparable to those for a traditional drainage system (Scholz *et al.*, 2006a), although maintenance costs for SUDS are on average 30% lower (Broad and Barbarito, 2004; Butler and Davies, 2000).

15.1.6 Water

Few studies on the costs of climate adaptation have been published, but many are likely to have been made (particularly in developed countries) as water supply is an important issue, but kept confidential for business reasons (Parry *et al.*, 2009). Parry *et al.* (2009) discusses two global scale studies of the costs of climate change in the water sector, and another world-wide study is reported here.

One study is Fischer *et al.* (2007) who investigates global and regional agricultural water demand for irrigation with and without climate change (following an A2 type scenario). Results indicate that climate mitigation could have a significant positive effect on water resources, by reducing impacts of climate change on irrigation requirements by 40% (or 125-160 billion m^3). Climate change could potentially mean costs for agricultural water withdrawals of US$25 billion by 2080, but climate mitigation combined with efficiency savings could cumulatively save US$10 billion by 2080. As time progresses, the net benefit of climate mitigation is greater, including in Europe. However, regional assessments indicate that not all global regions would initially benefit from climate mitigation and, therefore, the benefits and disadvantages, together with the associated costs, need to be carefully assessed (see Section 10).

Kirshen (2007) investigated adaptation and mitigation options with respect to water supply, and analysed this for seven global regions for 2030 (looking into a planning horizon of 2050) following the A1B and B1 scenarios. He looked at water resource availability for irrigation, agriculture, urban uses, domestic uses and industrial demands. Results indicated that demand growth at a national scale was extremely sensitive to costs under a climate change (A2) scenario. Kirshen (2007) also found the difference in capital costs between an A2 climate change and B1 mitigation scenario was US$4 billion between 2007 and 2030, with most of this occurring in developing countries. Due to the amount of overseas aid to help developing countries, with a changing climate, this value must be approximately doubled to meet the extra water production costs due to climate change. Adapting sources and supply of water in a changing environment helps to reduce these costs.

Finally, Hughes *et al.* (2010) analysed operating costs as a percentage of the adaptation costs of providing water infrastructures in OECD[8] countries. They looked at maintaining service standards under climate change up to 2050, but also took a longer term planning perspective until 2100. They considered the influence of climate change on water use and sewage, and drainage connections. A common pattern in their findings was that climate change tended to increase the amount of water use, but reduce the industrial demand in OECD countries, except in eastern Europe. The difference between these two sectors is due to temperature and precipitation patterns with respect to population density. As 60% of water is used in industry, there is a reduction in the volume of water required, but there could be a potential increase in costs due to treatment methods. Overall, the costs of adapting and maintaining the water infrastructure with respect to capital costs are around 1-2% across a range of climate scenarios, but with large regional variations.

At more local levels, adaptation is subject to a cost-benefit analysis, so that during the lifetime of an adaptation scheme, the benefits of controlling water must outweigh the financial costs. However, it does not allow non-financial benefits to be considered, e.g. ecological benefits and other indices, such as multi-criteria analysis, that can be evaluated to include cultural, societal and ecological effects or the efficiency of schemes.

On the demand side, managing and adapting people’s use of water can reduce usage. Despite the unit cost of water increasing over the past few decades, changing attitudes and actions of homeowners (e.g. metered water) have decreased the volume of water people use each day (European Environment Agency, 2012). To manage demand, public education and efficient water use are important.

15.2 Environmental impacts

Many of the adaptation and mitigation measures have environmental impacts other than those directly associated with the six sectors under consideration, or are relevant to several. These

[8] Organisation for Economic and Co-operation Development

include changes to soil properties, climate, air quality, water regulation and quality, and pests and diseases. This is particularly true in the case of changes to soil properties.

15.2.1 Soil

Mitigation in many of the other sectors involves various aspects of managing biodiversity and soils to enhance storage or prevent losses. This may be direct, e.g. managing soil organic carbon in agriculture (Section 4.1.1), or indirect as with the re-creation of wetlands to manage flooding (Section 3.6.2), which have the additional benefit of enhancing carbon storage.

In the case of forests, plantations can have increased nutrient demand affecting soil fertility and soil properties. Specifically, they can lead to higher erosion of the uncovered mineral soil surface (Perez-Bidegain *et al.*, 2001; Carrasco-Letellier *et al.*, 2004) and to significant changes of biological properties (Sicardi *et al.*, 2004) if the species selection does not take into consideration the site conditions. In numerous cases increased Na concentrations, exchangeable sodium percentage and soil acidity and decreased base saturation are observed in plantations (Jackson *et al.*, 2005; Brockerhoff *et al.*, 2007). Raulund-Rasmussen *et al.* (2011) demonstrated that various techniques like frequent logging, drainage, and soil preparation operations can lead to the depletion of the soil and humus carbon stocks. Moreover, management intensification (harvesting and site preparation) may decrease the soil carbon stock by as much as 50% through more frequent logging, drainage, and soil preparation operations.

15.2.2 Climate

As discussed in Sections 3.5.2 and 3.5.3, greenspace is an important way of adapting to climate change in urban environments, as it is able to influence the local climate; reducing local surface temperatures by shading, and reducing air temperatures through evaporative cooling and albedo effects (e.g. Gill *et al.*, 2007).

15.2.3 Air quality

Increasing urban greenspace, including urban trees, for example, can result in substantial improvements in air quality, by reducing particulate pollution, for example, PM_{10}. The capture of this particulate by urban trees in the Greater London Area has been estimated at approximately 852 tonnes per annum (Tallis *et al.*, 2011).

15.2.4 Water quality

Many studies in the water sector showed that pollution patterns may be altered due to changes in the hydrological cycle, as reduced water levels will mean that there is less dilution (Quevauviller, 2011). Pollution threats included soil salinisation, nitrogen loading, phosphorus loading, mercury deposition, pesticide loading, sediment loading, organic loading, potential acidification and thermal alteration (Vörösmarty *et al.*, 2010), and these factors can affect a large volume of water resources on a global to local scale.

One adaptation measure is the planting of vegetation to provide additional shade, so as to reduce water temperatures. It also provides the dual benefits of an additional habitat for wildlife and a carbon sink. This measure has been undertaken in response to the reported effects of increased temperatures in rivers and streams, which can reduce oxygen content and increase biological respiration rates leading to lower dissolved oxygen concentrations, particularly during summer months (European Environment Agency, 2007). Increases in temperature can also affect the habitats and the concentration of organisms, and some species may slightly shift northwards or experience small changes in their seasonal patterns (e.g. migration). There may also be changes to bacteria concentrations, nutrients and thermal stratification and water mixing in lakes.

15.2.4 Pests and diseases

In agriculture, a number of concerns have been raised about conservation agriculture practices (Soane *et al.*, 2012), for example, weed and pest control problems (Freibauer *et al.*, 2004; Sip *et al.*, 2009) and the widespread dependence of no-till on additional and regular applications of herbicides and pesticides. There is concern about their fate and environmental consequences, especially on water quality, although no explicit impacts were found as part of this review**.**

15.3 Social impacts

Few studies found in this review explored the social impacts associated with mitigation and adaptation options. Those which did often neglected to note that the measures were taken to minimise the risks of climate change. However, it can be assumed that the social impacts still apply here. Almost all cases of adaptation and mtigation in biodiversity, coastal and forestry sectors involve the creation of new habitat or greeenspace, having obvious amenity and recreational benefits. Although not commonly focused on in the literature, the main social considerations were found in discussion of coastal measures.

15.3.1 Coasts

Examples include studies by Thiere *et al*. (2011) who acknowledged the recreational benefit of nitrogen farming wetlands in southern Sweden and also by Luisetti *et al.* (2010; 2011), who note that managed realignment at the Blackwater Estuary, UK, would benefit groups, including birdwatchers and walkers. One coastal project which appears to have had an important social aspect is that of the Delta Project in the Netherlands where the Grevelingen Estuary was closed to create a saline lake (Saeijs and Stortelder, 1982). The demand for recreational opportunities at the lake was estimated by the study for day recreation (intensive – swimming and sun bathing; and extensive – enjoying nature); boating; fishing; and overnight recreation (e.g. tents and caravans) in the design stages of the project.

In contrast to habitat creation and naturalising the environment, Lamberti *et al.* (2005) acknowledged a social benefit from beach nourishment and the installation of groynes at a number of locations along the Italian coastline. This was important for visitors, the local tourism industry, as the wide beach makes the area more attractive to tourists (Lamberti *et al.*, 2005).

15.3.2 Agriculture

As far as agriculture is concerned, Bohlool *et al.* (1992) noted that socio-cultural constraints would limit the use of biological nitrogen fixation in agriculture as a form of climate change mitigation.

15.3.3 Urban

Similar recreational benefits were also created by urban measures such as greening and urban intensification among others. Many of these, however, were not considered specifically during the planning stages and, therefore, the review found few examples of where the social impact of adaptation-mitigation options was considered. Despite this, it was possible to find studies examining a social aspect, although they rarely noted the measures adopted as forms of adaptation or mitigation. In this section, the approach that similar social impacts will apply for adaptation and mitigation schemes is therefore taken.

As far as intensification is concerned, studies highlight that dense urban environments may be seen as unattractive by residents, who often prefer a less dense environment (Searle, 2010; Williams, 1999). Concerns over a loss of culture and local character were also given (Birrell *et al.*, 2005; Searle, 2010). Howley (2009) found that intensification in Dublin, Ireland, led to regions having younger, more affluent communities, than that of more established communities. The area saw an influx of young professionals to the area, with 8.3% of residents saying that the accessibility (26%), employment opportunities (12.8%), social life (5.8%) and cultural activities (2.5%) were the largest benefits from living in such a built-up area. Although densified urban environments are thought to increase social interaction, and it has been proposed that they strengthen feelings of community and safety, research shows that this is not always the case (Williams, 1999).

Studies from the US made a more explicit link with the social impact of adaptation/mitigation measures. A review by Pataki *et al.* (2011) noted the benefits of urban greenspace programs implemented for climate change included indirect improvements in human health, due to better air quality (a moderate effect, however, there exists much uncertainty here), and psychological benefits arising from the provision of cultural benefits, reduced stress, and a reduction in crime rates. Similarly, Clark *et al.* (2008) studying green roofs in Michigan, found these to improve human health; with fewer premature deaths and cases of chronic bronchitis.

16. Discussion

16.1 Method

This review has provided an indication of the range and importance of various sectoral adaptation and mitigation measures and their impacts. In many sectors, such as agriculture and water, there are a very large number of possible measures and so the research was focused on those which were most relevant to the CLIMSAVE modelling being undertaken in the IAP and the embedded adaptation responses and where there was adequate literature. This means that a large number of adaptation actions were not covered; a fuller coverage being provided by Berry et al. (2009). For example, for agriculture we did not examine aspects of pesticides and herbicide usage nor pasture or waste management for livestock. For other sectors, examples include expanding setbacks (e.g. distance between structures and shoreline), mapping coastal hazards (coastal flooding, cliff erosion risk for coasts), heat action plans, preparedness of health care systems, mapping of the urban heat island and cool places in urban areas, dams, re-naturalisation of rivers, groundwater recharge systems, and taxes or incentives such as those concerning amount of waste water used and water pricing for water.

16.2 Cross-sectoral interactions

The cross-sectoral impacts of adaptation and mitigation actions are not part of the IPCC chapters assessing the interactions between the two responses to climate change (e.g. Klein *et al.*, 2007), as they focus on the synergies, antagonisms (conflicts) and trade-offs, although some of the impacts of adaptation and mitigation are in the individual sectoral chapters. This review, found very few instances of neutral-no interaction (See Section 6), highlighting that there are many cross-sectoral impacts of adaptation and mitigation actions where the impacts can be both positive and negative for the given or another sector. It was difficult to find examples of simple cross-sectoral interactions, i.e. where impacts in the affected sector did not lead to any direct consequences for adaptation and mitigation actions in that sector. In contrast, the majority of cross-sectoral impacts from this review existed in the form of synergies and antagonisms. This interaction between mitigation and adaptation actions within and between sectors must to be taken in to account in any mainstreaming of adaptation (or mitigation) in sectoral policies to avoid unintended consequences (Klein *et al.*, 2007) or to enhance positive outcomes. This also was advocated by the EU White Paper on "Adapting to Climate Change"[9] and the recently adopted EU "Strategy on adaptation to climate change"[10]. The latter suggests that screening adaptation decisions will reduce the likelihood and frequency of maladaptation. This could result from negative impacts of an adaptation action on another sector, with or without affecting that sector's ability to adapt. It is recommended that all interactions, whether simple, synergistic or antagonistic, and trade-offs should be part of any formal assessment of the impacts of adaptation and mitigation measures. The importance of these findings will now be discussed.

16.3 Potential for synergies (win-win)

The review found that the greatest number of recorded cross-sectoral interactions impacted positively on both sectors, although there is a danger in assuming that the frequency of mention, or evidence of an interaction, represents the importance of a particular sectoral measure. More importantly it found that the effect on the impacted sector could often be considered consistent with adaptation measures for that sector, as shown by the italics in Table 14. This suggests that many synergies (and conflicts) are going unrecognised or are not being explicitly acknowledged and are under-represented in this review. Similarly past research has mostly failed to sufficiently evaluate synergies, with adaptation and mitigation often being considered independently (e.g. Klein *et al.*, 2007), and hence there is a need for future strategies to further explore the potential and maximise the benefits of synergies (Stoorvogel *et al.*, 2004).

The review also found that many of the positive cross-sectoral interactions involved biodiversity or water. This cross-cutting role of biodiversity has been highlighted by many stakeholders (Impact Assessment EU Paper on Adaptation). Many interactions with biodiversity involved habitat restoration or creation by other sectors (e.g. coasts), which can be considered generally positive for biodiversity, with such actions being part of many recommendations for biodiversity adaptation (e.g. Smithers *et al.*, 2008; Heller and Zavaleta, 2009). It is difficult, however, to judge whether these would always be undertaken for habitats which are particularly vulnerable to climate change and thus need adaptation actions to be taken, or are in locations where adaptation is needed (Berry, 2009b). Nevertheless, examples were found of where, for example, wetland creation increased carrying capacity for waterfowl, as well as creating suitable conditions for rare species (Wells and Turpin, 1999), including the endangered whooping crane (Hinkle and Mitsch, 2005; Darnell and Heilman, 2007). It was suggested in the coastal sector that these benefits once recognised, should be maximised during the design phase of similar coastal adaptation schemes, such as occurred in the managed realignment scheme at Wallasea, UK (Dixon *et al.*, 2008).

[9] http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2009:0147:FIN:EN:PDF
[10] http://ec.europa.eu/clima/policies/adaptation/what/docs/com_2013_216_en.pdf

The positive cross-sectoral interactions involving biodiversity could also potentially be considered to be part of ecosystem-based adaptation[11]. Potential benefits include biodiversity conservation, carbon sequestration, and sustainable water management. This type of adaptation is being promoted by the EU, for example, in the Strategy on adaptation to climate change and in the accompanying Impact Assessment, it is suggested that "there is growing recognition of the importance of ecosystem-based approaches by other sectors, particularly in relation to coastal protection, urban planning and water management" p33 (SWD (2013), 132). Such an approach also is stated in the EU biodiversity strategy to 2020 as a cost-effective way to address climate change adaptation and mitigation while offering multiple benefits beyond biodiversity conservation, which is where it has tended to be promoted. The cost-effectiveness of adaptation measures under climate uncertainty is being explored in CLIMSAVE through the use of various algorithms, in order that the cross-sectoral costs can be taken into account in the costs of any individual action.

This review found evidence of positive cross-sectoral interactions involving biodiversity in the form of green infrastructure, including green roofs, urban trees and sustainable urban drainage systems (e.g. Bowler *et al.*, 2010; Fioretti *et al.*, 2010). The EU White Paper on "Adapting to Climate Change - Towards a European Framework for Action (COM(2009), 147 final)", recognised that approaches such as this which work "with nature's capacity to absorb or control impact in urban and rural areas can be a more efficient way of adapting than simply focusing on physical infrastructure" (COM(2009), 147). Increasingly these ecosystem-based approaches to climate change adaptation and mitigation are being promoted, as biodiversity is seen as part of an overall adaptation strategy to help people adapt to or mitigate the adverse effects of climate change[12]. Ecosystem service frameworks would aid in the planning and evaluation of such schemes. This review has identified some of the synergies between biodiversity and the other sectors and these are examples of ecosystem-based adaptation or mitigation. It is interesting to note that biodiversity adaptation measures appeared to have little or no impact on the other sectors, whereas measures by the majority of other sectors were found to impact on biodiversity. Also, measures to adapt to future water resources by increasing resilience were often incorporated into strategies by sectors other than water, i.e. those which would be affected. For example, urban adaptation measures included RWHS, and measures for agriculture included reduced irrigation; both of which can be considered as water sector adaptation strategies, although the viewpoint in the literature (as seen) is often different. This overlap highlights the cross-sectoral nature of adaptation and mitigation.

[11] Ecosystem-based approaches to climate change adaptation (EbA) includes the sustainable management, conservation and restoration of ecosystems to provide services that help people adapt to the adverse effects of climate change. EbA often also contributes to climate change mitigation, by conserving carbon stocks, reducing emissions caused by ecosystem degradation and loss, or enhancing carbon stocks. (Report CBD AHTEG, 2009 and EC SWD (2013) 132, adapted).

[12] Connecting Biodiversity and Climate Change Mitigation and Adaptation - Report of the Second Ad Hoc Technical Expert Group (AHTEG) on Biodiversity and Climate Change under the Convention on Biological Diversity (CBD)

Table 14: Overview of adaptation interventions identified in this review, and their cross-sectoral interactions and effects on mitigation. Text in red indicates a negative interaction, and text in italics shows potential synergies.

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Agriculture							
Irrigation	Decreased supply to other water users; *water saving irrigation techniques could reduce demand*	Reduced water in rivers and lakes can adversely affect biodiversity, especially wetlands					Possible increase in soil C-storage; water saving techniques could reduce energy demand; intermittent irrigation of paddy rice can increase N_20 emissions
Crop type	Flooding	*Increase in water levels in wetlands*					
Earlier sowing dates	*Decreased water demand*/increase spring irrigation						Possible increase in soil carbon storage; spring sown crops could reduce N_2O emission
Breeding		Loss of genetic diversity					

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Conservation agriculture	*Improve crop water use efficiency; reduce N leaching, soil evaporation; increase water storage*. No-tillage can increase pesticide concentrations	*Increase soil fauna, including earthworm numbers; better habitat for micro-organisms*					Possible increase in soil carbon storage, decrease/increase in other greenhouse gas emissions depending on measure and its implementation
Biological nitrogen fixation	Reduced nitrogen leaching						Decrease nitrogen emissions
Targeting amount and timing of fertiliser application	Reduced nitrogen leaching						Decrease greenhouse gas emissions
Biodiversity							
Assisted colonisation	*New streams/wetlands increased water holding capacity & water quality; decreased flood risk*						
Corridors				Possible loss of forests	Possible loss of agricultural land		
Networks	*Peatland restoration increases water storage, reducing flooding*				Possible loss of agricultural land	Reduced sediment supply	

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Habitat restoration	*Can reduce flooding*						Restoring wetlands will increase carbon sequestration
Protected areas				New PAs could lead to loss of forest	New PAs could lead to loss of agricultural land		
Coasts							
Wetland creation	Altered soil redox potential; Long-term improvement in water quality; short-term may be negative	*Wetland habitat creation/restoration; increased species richness and carrying capacity*			Loss of agricultural land		Increase carbon sequestration; increase in CH_4 and N_2O emissions
Managed realignment	Long-term improvement in water quality; short-term may be negative	*Habitat creation/restoration; benefits most species*	*Increase*/decrease urban protection	Loss of forest	Loss of agricultural land		Increase carbon sequestration; increase in CH_4 and N_2O emissions
Managed retreat	Possible short-term reduction in water quality followed by overall improvement	*Habitat creation; benefits most species*	*Increase*/decrease urban protection		Loss of agricultural land		Increase carbon sequestration; increase in CH_4 and N_2O emissions

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Low crested structures		*Provision of novel habitat; fish nursery ground; increase in algae*, but can prevent species settling on structure. Coastal squeeze					
Beach nourishment		Loss of dune vegetation; loss of species					
Storm surge barriers	Improve water clarity	*Improved water quality can increase phytoplankton productivity. New habitat can be created behind the barriers.* Can destroy/degrade ecosystem, e.g. tidal flats	*Protection from flooding*				Tidal barriers if combined with energy production could reduce fossil fuel demand; lakes behind them can increase local temperatures; loss of habitat
Forests							
Afforestation/reforestation	Reduced river flow, groundwater recharge. Planting on agricultural land can restore water quality	*Can improve/increase diversity;* habitat loss/change; species loss due to chemical inputs & forest management			Loss of agricultural land. Conversion of land or intensification of farming elsewhere		Increased storage on newly planted land

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Urban							
Green roofs	*Stormwater, infiltration and flow reduction.* Drainage ditches increase peak flows in early stages of plantation	*Habitat provision,* but challenging environment					Carbon sequestration (small); reduce energy demand through decreasing temperatures
Urban trees and greenspace	*Runoff reduction.* Can reduce air quality, emitting BVOC and aiding the formation of smog						Carbon sequestration; reduce energy demand through decreasing temperatures
White-topping/cool pavements	*Reduced stormwater loadings, improvement in infiltration, water retention and evapotranspiration, decreased water demand from other sources*						Reduce energy demand through decreasing temperatures
Rainwater harvesting	*Reduces household demand, decentralises water supply*				Intensification can protect agricultural land from development		

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Building measures	*Reduce amount and peaks of runoff and flooding*, improve water quality	*Restore certain ecosystem functions;swales and ponds increase habitats*					
Sustainable urban drainage systems (SUDS)	*Reduces runoff, aids stormwater retention. Pervious pavements filter and store runoff, improving water quality via reducing diffuse pollution in urban watercourses*	*Can provide habitat*					
Water							
Increased infiltration e.g. changing tillage practices; storm water control			*Reduce urban flooding*		*Increase soil water availability*		
Increased storage e.g. reduced drainage; RWHS afforestation; wetland restoration		*Ponds can increase biodiversity*			Reduced sediment supply; saline intrusion		Ecosystem-based measures could increase carbon sequestration
Reduced flood impact e.g. defences, planning, floodplain restoration					Could reduce water availability depending on prioritisation of use		Ecosystem-based measures could increase carbon sequestration

	Sectors impacted by adaptation measure						
Adaptation and mitigation actions	**Water**	**Biodiversity**	**Urban**	**Forests**	**Agriculture**	**Coasts**	**Mitigation effect**
Flood plain restoration	Improve water quality	*Increase in wetland habitat and species*					Ecosystem-based measures could increase carbon sequestration
Reservoirs		Changed biodiversity. Loss of riverine species					Direct increase in greenhouse gas emissions; reduce emissions from fossil fuel if used for HEP

16.4 The wider impacts of interventions

There is the possibility also that while the positive cross-sectoral interactions of adaptation or mitigation measures in one sector may not contribute to adaptation or mitigation in the other sector, nevertheless they can improve environmental conditions, such as water and soil quality, in the impacted sector. Measures such as these increase adaptive capacity by increasing resilience and robustness both to climate and other changes (Smith *et al.*, 2001; Tol, 2005). They are, therefore, often seen as low, or no-regret, as the benefits from these measures are realised regardless of the uncertainties surrounding future climate projections (Hallegatte, 2009). For example, this review found evidence of earlier sowing dates being adopted in agriculture to reduce drought stress (e.g. Moriondo *et al.*, 2010) – a problem which already exists irrespective of future changes and, hence, this adaptation will provide current, as well as future, benefits. Similarly, in urban areas the implementation of rainwater harvesting and greywater re-use decentralise water supply, reduce potable water use, and increase regional resilience to drought by improving security (Graddon *et al.,* 2011). In the absence of synergies, such actions should be preferred, as they are likely to produce overall environmental benefits and be more cost-effective.

It is logical to state that strategies involving a high number of synergies (both for mitigation and adaptation) should be favoured. However, aspects such as the flexibility of schemes, the extent to which they offer no-regret solutions and increase resilience, are also important to consider (Adger *et al.*, 2005; Hallegatte, 2009). From this viewpoint the impact of climate change uncertainties can be substantially reduced. An assessment of adaptation options examined in this review is given in Table 15, which ranks preference that could be given to various strategies. It is important to consider a number of factors when conducting such an assessment. For example, although habitat creation and wetland creation both have synergies with mitigation, the latter is known to be a very effective carbon sink (e.g. Choi *et al.*, 2001; Trulio *et al.*, 2007), whereas the extent of mitigation provided by habitat creation is highly dependent on habitat type. Similarly, the strength of mitigation provided by genetic modification in agriculture, as opposed to afforestation with climate-resilient genotypes, depends on the ability of new species to sequester carbon. Forests are known to store large amounts of carbon, but the ability of modified crops could be substantially less (e.g. Peoples *et al.*, 1995; Bonesmo *et al.*, 2012). Taking such factors into consideration, it appears that some of the most favourable options are those which work across sectors, building on the natural capacity of biodiversity to provide ecosystem services. For example, SUDS options and green infrastructure options benefit adaptation in the water sector and in urban areas as well as contributing to mitigation through carbon storage and providing habitat for biodiversity.

Table 15: Adaptation measures for the sectors and their assessment compared to other strategies found in this review. For the no-low regrets, "++" indicates measures that will produce benefits regardless of climate change, "+" indicates no-regret in some cases, depending on circumstance. Ranking of measures: rank 1 being the most favourable options and 3 the least. Favourable options will be those with synergies which increase resilience and are hence no-low regret. The least favourable options will be those which involve a number of conflicts, and/or are not flexible, based on findings from the review. Table adapted from Hallegatte (2009).

Sector	Examples of adaptation options	No-low regret	Reversible / flexible	Synergies with mitigation	Synergies with adaptation in other sectors	Ranking
Agriculture	Changing planting dates	+	+			2
	Genetic modification		-	+		3
	Conservation agriculture	+		+	+	2
	Development of more resistant crops	+		+		1
Urban	Green infrastructure	++	+	+	++	1
	Building measures	++	-	+		2
Water	SUDS	++		+	++	1
	Flood defences		--		+	3
	Storage	+				3
	Floodplain restoration	+		+	+	2
Biodiversity	Corridors and networks	++	+	+	++	1
	Restoration schemes	++		+	+	2
	Habitat creation	+		+	+	3
Forest	Chemical control for pests and disease		-			3
	Afforestation with climate-resilient genotypes	+	-	++	+	1
Coastal	Hard-engineering		--	-		3
	Managed realignment	+		+	+	1
	Wetland creation	+		++	++	1

16.5 Antagonisms and trade-offs

In contrast to the above positive interactions, negative ones were also identified and these, like the conflicts which will be discussed below could lead to the need for trade-offs. The interactions were mostly concerned with water quantity and quality, biodiversity and competing land uses. The hard engineering adaptation approaches in the coastal sector were especially linked to a number of antagonisms with biodiversity as a result of coastal squeeze (Bozek and Burdick, 2005). In addition to antagonisms, a number of examples were found of measures with negative impacts on other sectors.

The number of European adaptation and mitigation strategies for which trade-offs can be identified (whether implicitly or explicitly) goes to highlight the importance of more integrated management. These include for the agricultural sector: the uptake of deficit crop irrigation despite a potential reduction in yield (Mushtaq and Moghaddasi, 2011); reductions in CH_4 emissions from manure stores constrained by increased energy emissions required for cooling (Dalgaard *et al.*, 2011); and the creation of nitrogen farming wetlands despite the potential for CH_4 emissions (Thiere *et al.*, 2011). Furthermore, existing conflicts between irrigation, public water supply and environmental protection lead to the need for the consideration of a range of trade-offs (Daccache *et al.*, 2012). A good overview of some of the complexities of trade-offs in this sector is given by Herrero *et al.* (2009). Numerous trade-offs are also present in long-term coastal management, however, these can be overcome by the development of a more coherent cross-scalar approach to planning in addition to increased collaboration during the decision-making process (Few *et al.*, 2004).

Integrated city models have been identified as a tool with which to identify trade-offs and synergies between urban climate policies (Viguié and Hallegatte, 2012). Policies of (1) greenbelt, (2) zoning to reduce flood risk, and (3) a transportation subsidy were examined, with the model finding that the only way to achieve a win-win outcome was to combine the policies rather than to develop them separately. However, it is important to note that sometimes trade-offs are unavoidable (Viguié and Hallegatte, 2012). In such instances, taking findings from the coastal sector, informing stakeholders and the public of trade-offs is vital to gaining their support in long-term coastal planning (Tompkins *et al.*, 2008).

It is interesting that although the review found many examples of adaptation and mitigation strategies being implemented in Europe, the majority failed to consider the ‘success’ of actions. This may be a result of the complexities involved, for example, success is dependent on the interaction of adaptation and mitigation measures over time, whether this be cross-sectoral, synergistic or antagonistic (Adger *et al.*, 2005). A very limited number of studies considered the impact of actions over the long-term. Furthermore, success will be influenced by future conditions (climatic and otherwise) which are surrounded by uncertainty.

16.6 Dealing with conflicts

International, national and local policies are one way to reduce conflicts and enable the resolution of trade-offs. This will be illustrated with regards to biodiversity as this is the sector affected most by actions in others. The EU Habitats Directive 1992, for example, states that for any new development where wetlands are lost, new wetlands are required to be created to compensate for this. However, original and newly created wetlands absorb carbon at different rates, and thus in a new (compensated area) development there is the potential for the net loss of carbon storage. Although there is still uncertainty regarding how much carbon is lost, Hossler and Bouchard (2010) suggest, based on US studies, that a compensation ratio of 5:1 for the area of new wetlands should be created against the original wetland area to ensure that the same volume of carbon is stored. Van Roon (2012) builds on this argument as she believes that peatlands are an undervalued resource that could be used for dual benefits to store carbon, mitigate the effects of urban development and increase biodiversity. Thus, natural rural peatlands can do this in the most effective manner, as opposed to created wetlands in urban areas.

While not explored as part of the CLIMSAVE literature review, another possible way of dealing with some conflicts from a biodiversity viewpoint is biodiversity offsetting. These are market-based schemes which seek to "offset" damages caused by development, such that at a minimum, there is no net loss of biodiversity and preferably some gains. A number of European countries are implementing measures for biodiversity offsetting, with a good reviews given by ten Kate *et al*. (2004) and Quétier and Lavorel (2011). A pan-European mechanism is that of the Natura 2000 network, whereby to comply with the Habitats Directive any development which will adversely affect the network must be offset by conservation measures elsewhere in the network (ten Kate *et al*., 2004). Individual countries are also employing their own mechanisms for biodiversity offsetting, with those of 'Ausgleich' and 'Biotopwertverfahren' in Germany (Bruns, 2007), 'Offset Ratios' in France, and federal law in Switzerland to protect the country's nature and landscape[13]. It is clear however, that measures employed in existing biodiversity offsetting schemes are ineffective and need to improve substantially, considering a large number of factors and habitat equivalence (Walker *et* al., 2009; Quétier and Lavorel, 2011).

16.7 Barriers

There exist a number of barriers to the effective development and implementation of adaptation and mitigation schemes (for good examples see Ivey *et al.*, 2004; Crabbé and Robin, 2006). These include current policy approaches, institutional complexities, action over a wide range of scales, insufficient information and communication (Howden *et al.*, 2007; Biesbroek *et al.*, 2010). Burch (2010a; b) developed a framework for exploring the institutional and behavioural barriers to climate change policy development and implementation. Building on this, work on the barriers to embedding climate change adaptation principles for biodiversity identified the following categories of barriers: governance and leadership, institutional structure, legislative/policy history, relevance, capacity, conflicting priorities and reluctance to change (Berry *et al.,* 2010). Interviews with key individuals involved in the different workstreams of the England Biodiversity Strategy and other delivery partners found that the key barriers included: uncertainty about both the future of funding and climate change as a policy priority, organisational silos resulting in insufficient communication of the relevance of adaptation to conservation, and policy legacy leading to sub-optimal outcomes under climate change (Burch and Berry, 2013).

Portugal, a country whose coasts are at risk from coastal flooding and erosion, provides a good example of where effective coastal planning has been prevented by the presence of a number of barriers (Schmidt *et al.*, 2012). These include shortcomings in the clarity of policies and political support, insufficient finances, limited integration of knowledge and weak coordination of stakeholders. These barriers led to problems in organising the institutions and stakeholders involved, with many discrepancies in policies, and responsibilities which overlapped between plans and coastal planning legislation (Lopez-Alves and Ferriera, 2004; Carneiro, 2007; Schmidt *et al.*, 2012).

[13] www.admin.ch/ch/f/rs/451/a18.html, Accessed: June 2013

16.7.1 Policy

Policy instruments, both soft (e.g. financial incentives and voluntary agreements) and hard (e.g. regulatory measures or sanctions) can aid the implementation of adaptation measures, although little evidence has been found of these being developed (Biesbroek *et al.*, 2010). This research has found limited evidence for the use of policy tools; one example of which was tax incentive programs in the urban sector to encourage development on brownfield sites (Bunce, 2004; Hayek, 2010). The potential impact of an emissions tax on agriculture in south-west Germany was also assessed (Neufeldt and Schäfer, 2008), as well as the impacts of agricultural policies such as CAP and subsidies (Freibauer *et al.*, 2004).

Schemes, such as the corridors forming the Natura 2000 network, the European Greenbelt (Zmelik *et al.*, 2011), and the low energy residential settlement of Borgo Solare (Aste *et al.*, 2010), were implemented under or comply with a number of policies, including the Water Framework Directive, the Bird and Habitat Directives, the EU Floods Directive and the European Directive on Energy Performance of Buildings. It is interesting to note that some policies which impact nature conservation, such as the Natura 2000 legislation, are seen as posing barriers to successful adaptation (although there is some debate about its flexibility for dealing with climate change impacts e.g. Beunen, 2006). For example, strategies targeting biodiversity need to address changing climatic conditions and the shifting ranges of species, and hence need to be flexible. In contrast, the Natura 2000 policy is viewed as more rigid, causing institutional difficulties for habitat offsetting and banking schemes. A further problem is that climate policy, for example that for the urban sector, interacts with other targets including economic competitiveness and social issues (Viguié and Hallegatte, 2012). The broad reach of climate change interventions means that they are far from being independent of other decisions (O'Brien and Leichenko, 2000). It can hence be difficult to coordinate approaches to meet all targets.

Despite the clear existence of climate change policies, it has been found for many EU countries that these are not incorporated explicitly into existing sectoral policies (Urwin and Jordan, 2008; Biesbroek *et al.*, 2010). Climate change needs to be better integrated within current and future policies (on a national and EU level), rather than being dealt with separately (Howden, 2007; Biesbroek *et al.*, 2009; EU White Paper). This is key, as it would allow the identification of the impacts that measures taken under one policy will have in another policy domain (Biesbroek *et al.*, 2009). It has been highlighted that if urban climate policies, such as zoning to reduce flood risk, transportation subsidies and a greenbelt policy, are considered and implemented individually, they are likely to be unacceptable from a political viewpoint as they adversely affect each other, however, when policies are implemented in a mix, there is potential for acceptable win-win strategies (Viguié and Hallegatte, 2012).

Successful strategies will require assessment frameworks to be implemented, which are robust, relevant and easily accessible to stakeholders, policy-makers and the scientific community (Howden *et al.*, 2007). Bottom-up approaches to policy-making, although complex have been found to highlight cross-sectoral interactions, whereas there are cases where a top-down approach can lead to unintended antagonisms (Gupta, 2007; Urwin and Jordan, 2008). Existing policies therefore need to be reviewed and altered to assist the adaptation process (White paper). An example of policy improvement is seen in Portugal's Finisterra programme. This was implemented in 2003 and consists of a coherent framework law on coastal areas, and utilises numerous legal tools such as the creation of a supra-ministerial coordination body for coastal management (Schmidt *et al.*, 2012).

16.7.2 Knowledge

As stated previously (see Section 1), adaptation and mitigation measures are interlinked, making it important to seek inter-disciplinary solutions, having a strong exchange of ideas and information with decision-makers (Howden *et al.*, 2007). The numerous synergies, antagonisms and cross-sectoral impacts identified in this review support the need for such an approach. It is, therefore, not possible to examine only mitigation or adaptation alone, nor is it possible to consider only the sector for which the measure is intended. Findings from this review reflect the need to overcome barriers towards achieving this goal.

The potential for feedbacks, i.e. synergies and trade-offs, will need to be assessed using a holistic and fully integrated framework approach, which examines a number of sectors (Jarvis *et al.*, 2011). This would allow for the cross-sectoral management of adaptation and mitigation options (Howden *et al.*, 2007). Modelling of impacts, therefore, also needs to be integrated, incorporating socio-economic, biological and physical factors, such as has been undertaken in the CLIMSAVE project. The identification of key impacts and metrics for cross-sectoral comparison has also been part of CLIMSAVE and this will be able to contribute to the assessment of the effectiveness of different adaptation measures across sectors.

EU members need to better coordinate knowledge within countries, for example, improving the communication of climate information between the large number of organisations contributing research and knowledge; including national governments, meteorological institutions, research institutes and programs, NGOs and special organisations (e.g. the Danish Information Centre on adaptation, established under the Danish National Adaptation Strategy) (Biesbroek *et al.*, 2010). It will be extremely important to share this knowledge between the scientific community, policy and decision-makers, practitioners etc. to improve adaptation-mitigation recommendations (Jarvis *et al.*, 2011; Clar *et al.*, 2013). CLIMSAVE has gathered evidence at a European scale and experience from past schemes can aid planners. Public awareness also needs to be increased via a number of tools, such as the CLIMSAVE IAP (Harrison *et al.*, 2012), and practitioners need to be made aware of the full range of suitable adaptation measures available (Biesbroek *et al.*, 2010).

Some of the strategies examined in this review have further highlighted the lack of long-term monitoring and assessment of their impacts, this only being conducted for very few schemes. Such as the results from this review, many National Adaptation Strategies also fail to acknowledge the importance of monitoring and how it should be undertaken (Biesbroek *et al.*, 2010). The UK and Finish strategies are among the only few which discuss the development of quantitative indicators to assess effectiveness (Swart *et al.,* 2009; Biesbroek *et al.*, 2010). Despite the inadequacy of current monitoring, it is essential to measure the effectiveness of strategies (Magurran *et al.*, 2010; Huntjens *et al.*, 2012). Metrics concerning the effectiveness of mitigation schemes are more developed than those for adaptation (e.g. van Minnen *et al.*, 2008), despite debate over the amounts of carbon stored in various ecosystem components. Clearly a suitable framework and selection of tools needs to be developed to evaluate strategies. Solutions could include the evaluation of schemes using environmental impact assessment procedures (Agrawla *et al.*, 2012; Wende *et al.*, 2012), or frameworks to assess the resilience of options (Engel *et al.*, 2013).

Although the communication of knowledge and monitoring are barriers which can be successfully removed, the lack of certainty of climate change and its impacts proves much more challenging (Agrawla *et al.*, 2012). Detailed information required for successful evaluation of schemes and their impacts is currently unavailable, or is associated with a high number of uncertainties (Hulme *et al*., 2007; Moser, 2009; Smith *et al*., 2009; Clar *et al*., 2013). For example, local scale climate projections which would be appropriate for decision-making at the project level are, however, associated with a high number of uncertainties (Agrawala and van Aals, 2005; Wilby and Dessai, 2010). In the agricultural sector, some progress has been made with the development of ensemble agriculture and climate models to examine adaptation options (Challinor *et al*., 2013).

16.7.3 Governance and actors

Despite the number of cross-sectoral interactions found in this review, few studies discussed how schemes were managed. Many of the examples found tended to be quite specific to the regional or national scale, posing a barrier to solutions and the successful coordination across levels. A number of studies from this review did, however, highlight the vast array of actors and governance levels involved. For example, the de Doorbraak project in the Netherlands required communication across *district* water boards, the *local* province, and the *national* Ministry of Agriculture, Nature Management and Fisheries (WRD, 2011). Again in the coastal sector, examples were found of partnerships across a number of organisations, with actors involved in the managed realignment at Hesketh Out Marsh West, including a UK *government agency*, an *NGO* and the *local counc*il (Tovey *et al*., 2009). Information on governance for the coastal sector shows that national levels of governance are often employed although the majority of plans are local, e.g. managed retreat at Orplands, UK (Emmerson, 1997). In the urban sector, examples of local and regional building code regulations were seen in the Sant Cugat del Vallès municipality and the region of Catalonia (Domènech *et al.*, 2011). Literature covering the agricultural sector provided examples of the EU-level of governance, with EU frameworks concerning organic farming (e.g. Framework of Organic Food and Farming, EC 834/2007; the European Organic Seed Regulation, EC 1452/2003). The above highlights examples of multi-level governance. Government regulations and policies implemented at the European level need to be downscaled in order to form policy strategies and targets at a local level. Such interaction between governance levels and participatory approaches involving stakeholders and institutions are required to overcome complexity issues (Biesbroek *et al.*, 2009).

In addition to the above complexity of actors, CLIMSAVE results highlight the trans-boundary nature of some schemes. The ability, and perhaps the need for EU member states to work together was best highlighted by the Wadden Sea Plan: a trilateral cooperation between Germany, the Netherlands and Denmark; aiming to re-naturalise the coastline and conserve biodiversity (Falk, 2004; Enemark, 2005). Another international cooperation is the Restoring Peatlands Project in Belarus, which is a result of research and experience from experts in Belarus, Germany and the UK. The scheme is financed by Germany as part of the International Climate Protection Initiative by its Ministry for the Environment, Nature Conservation and Nuclear Safety. Further increasing the complexity of actors, the project is coordinated by the UK's Royal Society for the Protection of Birds in collaboration with APB-BirdLife Belarus and the German Michael Succow Foundation. The scheme is further supported by the UN Development programme in Belarus and the Ministry of Natural Resources and Environmental Protection of the Republic of Belarus[14].

Studies in this review mostly failed to explore the role of National Adaptation Strategies (NASs) in adaptation and mitigation schemes, despite these being adopted by 15 EU member

[14] http://www.restoringpeatlands.org, accessed 20/08/2012

states[15]. These aim to better facilitate adaptation across a range of spatial and temporal scales, and to promote governance at multiple levels (Biesbroek *et al.*, 2010). NASs differ between countries, although many of the drivers (e.g. EU policies), and methods and approaches used are similar (Swart *et al.*, 2009; Biesbroek *et al.*, 2010). As found in this review, cross-sectoral impacts are numerous and it is important to recognise these to maximise the benefits of schemes and avoid conflicts. In contrast, research finds that not all European NASs identify this link (Biesbroek *et al.*, 2010). Table 16 highlights that many country's plans are not considering the cross-sectoral nature of schemes, although this review found biodiversity to be a cross-cutting issue, it was only shown in the two out of the seven NASs examined. The Spanish NAS (PNACC, 2006) does, however, recognise that water, biodiversity and coastal sectors can impact on agriculture and forestry; whereas the French NAS (ONERC, 2007) actively identifies cross-cutting issues, such as water and biodiversity; and sectoral approaches such as agriculture. It is interesting to note that the Belgian NAS is the only one which explicitly considers the cross-boundary aspect of climate change in adaptation strategies (Swart *et al.*, 2009).

Table 16: Sectors involved in selected adopted National Adaptation Strategies. Germany (DE), Denmark (DK), Spain (ES), Finland (FI), France (FR), the Netherlands (NL), the UK, Hungary (H), Portugal (P) and Belgium (B). Two crosses mark priority sectors or cross-cutting issues for some countries. Table adapted and updated from Biesbroek *et al.* (2010).

Sector	**DE**	**DK**	**ES**	**FI**	**FR**	**NL**	**UK**	**H**	**P**	**B**
Agriculture	X	X	X	X	X	X	X	X	X	X
Biodiversity	X	X	XX	X	XX	X	X		X	X
Forests	X	X	X	X		X	X	X	X	X
Coasts	X	X	XX			X	X		X	X
Urban	X	X		X		XX	X	X	X	X
Water	X	X	XX	X	XX	XX	X	X	XX	XX

16.7.4 Scale

This study highlighted the variation in scale of adaptation measures between sectors (see Section 12). Table 17 shows the contrasting spatial scales of adaptation measures for each sector. Instances of international strategies were found mainly in the agricultural, biodiversity and coastal sectors. Biodiversity adaptation occurred at multiple scales, often supported by EU-policies and strategies, including the Convention on Biological Diversity, the Pan-European Biological and Landscape Diversity Strategy[16], the European Green Belt (Zmelik *et al.*, 2008) and European ecological networks such as the Emerald Network[17], which highlight the trans-boundary nature of adaptation in this sector. Coastal adaptation was often local in scale, inferring the site-specific nature of adaptation requirements here, although the UKs intertidal agri-environmental scheme is one example of adaptation at the national level (Parrott and Burningham, 2008). In contrast to these international scale examples, urban was the only sector in this review where adaptation measures were not implemented at a scale greater than regional, with many options taken at the household level (e.g. Benemann and Chebab, 1996), however, local adaptation measures can impact at higher levels, for example, by increasing regional resilience to drought.

[15] http://www.eea.europa.eu/themes/climate/national-adaptation-strategies, accessed 24/06/2013
[16] http://www.peblds.org/, accessed: 24/06/2013
[17] http://www.coe.int/t/dg4/cultureheritage/nature/econetworks/default_en.asp, accessed: 24/06/2013

Previous studies have often highlighted that adaptation and mitigation concern action on different scales, with adaptation being mostly achieved at a local, small scale; whereas mitigation is an international issue, dealt with by action from national governments and in international agreements (Tol, 2005; Biesbroek *et al.*, 2009; Jarvis *et al.*, 2011). In contrast to this mismatch of scales, this review found that schemes were implemented at similar scales. For example, mitigation actions such as tree planting (Davis *et al*., 2011), green roofs (Getter *et al*., 2009) and low energy residential developments (Chance, 2009) in urban areas; and local saltmarsh and floodplain restoration schemes, and conservation agriculture (Six *et al*., 2004), are all implemented at small scales – often local. Adaptation options, such as SUDS (Andersen *et al.*, 1999), building measures (Artmann *et al*., 2008), testing genetic diversity (Singh and Reddy, 2011), changing seed sowing dates (Tubiello *et al*., 2000), and the construction of LCS (Lamberti *et al*., 2005) again all occur at local scales. It seems, therefore, that adaptation and mitigation actions occur at very similar, local scales (Wilbanks and Kates, 1999; Schreurs, 2008). This is not to say that local projects will individually achieve reductions in atmospheric GHG concentrations, or to neglect the fact that some mitigation projects are much larger in scale, however, from this review mitigation actions seem to be implemented from the bottom-up: the end result being the collective impact of local efforts driving climate change mitigation in Europe. Mitigation has rarely been considered in this way, with only a few authors examining this theory (e.g. Wilbanks and Kates, 1999; Lutsey and Sperling, 2008).

Table 17: Spatial scales of sectoral adaptation measures.

<table>
<tr><th></th><th colspan="4">Spatial Scale</th></tr>
<tr><th>Sector</th><th>Local</th><th>Regional</th><th>National</th><th>International</th></tr>
<tr><td rowspan="2">Agriculture</td><td rowspan="2">Mostly local, e.g. testing of genetic diversity</td><td rowspan="2"></td><td></td><td>Development of new genes (Ortiz, 2008)</td></tr>
<tr><td colspan="2">Development of new cultivars</td></tr>
<tr><td rowspan="2">Biodiversity</td><td rowspan="2">Habitat restoration</td><td rowspan="2">Assisted migration</td><td></td><td>The CBD, Bird and Habitat Directives.</td></tr>
<tr><td colspan="2">Networks and corridors</td></tr>
<tr><td>Urban</td><td>Household-neighbourhood scale.</td><td colspan="2">Building regulations</td><td></td></tr>
<tr><td>Water</td><td>Irrigation, RWHS, SUDS</td><td></td><td></td><td>EU Habitats Directive, WFD</td></tr>
<tr><td>Forestry</td><td>Afforestation, restoration</td><td></td><td></td><td></td></tr>
<tr><td>Coasts</td><td>Mostly local, e.g. restoration, managed realignment</td><td></td><td>UK intertidal agri-environmental scheme</td><td>The Wadden Sea Plan</td></tr>
</table>

Differences in temporal scale for adaptation and mitigation were also seen in this review, although evidence was lacking for many of the measures for biodiversity adaptation and mitigation. Mitigation actions often led to long-term benefits, and near-term benefits were achieved by adaptation measures, as found by numerous authors (e.g. Dessai and Hulme, 2007). Many adaptation measures found in this review, such as changes to sowing times, building measures, and RWHS can be implemented (relatively) quickly (Trnka *et al.*, 2004).

In contrast, the review also found evidence of adaptation occurring over much longer timescales, for example, the creation of ecological networks and new protected areas to deal with species migration in response to climate change, and afforestation using more climate-resilient genotypes (FAO, 2010). In a similar way, many mitigation efforts, for example, the creation of saltmarsh for carbon storage in the coastal sector (Choi *et al.*, 2010), or reforestation in the forestry sector for carbon sequestration purposes (Ding *et al.*, 2011), take place over a much longer timescale and require longer to become effective. These findings show that in addition to the false perception of a mismatch of spatial scale, there are also instances, where the temporal scale of mitigation and adaptation measures can be similar. Past literature has often emphasised the temporal and spatial mismatch of scales as posing a barrier to the integration of mitigation and adaptation, and the successful evaluation of trade-offs (Tol, 2005; Howden, 2007). Results from this review, however, suggest that there are many cases in which the scales are comparable, hence, removing one of the perceived barriers.

As far as management is concerned, it will be important to ensure that any short-term adaptation measures do not prevent or hinder longer term adaptation or mitigation options, and therefore the impacts of actions need to be considered across a range of timescales and across the lifetime of schemes to examine their wider impacts (Adger *et al.*, 2005).

16.7.5 Institutional complexities

None of the studies in this review gave much attention to the institutional difficulties associated with adaptation or mitigation schemes, however, the above results do highlight the importance of successful communication and interplay between sectors, actors and governance. The various actors, institutions and organisations identified as being involved in adaptation and mitigation schemes have contrasting complexities, and importantly they often have a limited experience in working together. For example, adaptation in the water sector in Mediterranean countries has displayed a rather limited cooperation between institutions, and coordination between states, administrative regions and river basin authorities has been rather disjointed (Iglesias *et al.*, 2007). The importance of collaboration among actors has been highlighted, and it has been shown that a strictly sectoral approach could result in a number of conflicts and trade-offs with other sectors. For example, if coastal flood management takes a purely sectoral approach, hard engineering measures are likely to be employed which are likely to impact negatively on biodiversity (e.g. Smits *et al.*, 2006). Under a more coordinated approach among sectors, soft management schemes, such as managed realignment are more likely to be favoured, with benefits for other sectors. This does, however, call for flood and coastal erosion risk management strategies to take into account the interests of, for example, agriculture and biodiversity conservation.

For adaptation to be successful, public and private actors will need to collaborate across all levels of governance (from international-national to local) making this an issue of multi-level governance (Biesbroek *et al.*, 2010) which needs to be well coordinated across sectors (Biesbroek *et al.*, 2009; Jarvis *et al.*, 2011). Research highlights the importance of institutional frameworks to aid in strengthening the adaptive capacities of individuals such as farmers (Jarvis *et al.*, 2011). The Netherlands provides a good example of how such multi-level governance can be applied with their national programme created to have representatives at the national, provincial and municipal levels, in addition to representatives from, for example, water boards and scientists to discuss the complex dimensions of adaptation interventions (Biesbroek *et al.*, 2010). One of the most effective specialist organisations coordinating research between science and policy is the UK Climate Impacts

Programme (UKCIP) which was established in 1997 (West and Gawith, 2005). The programme adopts a fully integrated approach to provide advice on the creation of adaptation policies for decision-makers through the provision of a range of stakeholder tools and climate impact studies. In contrast, in terms of organisational structure in the UK for embedding adaptation policies in, for example, biodiversity, there are some important obstacles which need to be overcome (Berry *et al*., 2010). For example, the Department of Energy and Climate Change (DECC) is an entirely separate institution to the Department for Environment, Food and Rural Affairs (DEFRA). This separation creates silos between adaptation policies and biodiversity. Further institutional difficulties include the fact that as a result of their structure, not all organisations have the appropriate knowledge of actions being taken (Berry *et al*., 2010) For example, those working loosely with the England Biodiversity Group have very limited knowledge of the activities undertaken by organisations, such as Defra. Such an organisational structure is clearly inefficient and could lead to conflicting actions being taken. It is therefore essential that institutional complexities, such as that above are overcome, to improve the efficiency and success of future schemes.

16.8 Future outlook

This discussion has found that a number of barriers exist which hinder the development and implementation of adaptation and mitigation measures. This issue is far from small and it has been suggested that institutional complexities, such as policy integration and multi-level governance, could be more of a challenge to adaptation than the creation and identification of suitable technical solutions (Biesbroek *et al.*, 2010). It is, therefore, apparent that current approaches need to improve and institutional difficulties especially need to be overcome. This review has also highlighted the need for integration between mitigation and integration to avoid maladaptation, and to promote win-win with maximum benefits and efficiency.

In order to put this into practice, it may be best to focus on the biodiversity and water sectors as there were a high number of synergies associated with actions here. Also, adaptation and mitigation measures, both within these sectors and impacting on these sectors, were shown to have a cross-cutting nature. To some extent the importance of biodiversity is already being recognised through the development of policy relating to the promotion of ecosystem-based adaptation and the EU strategy for promoting green infrastructure (COM (2013), 249). Such strategies use biodiversity to enhance natural ecosystem services and aid adaptation through increasing the health and resilience of ecosystems (Kazmierczak and Carter, 2010; COM (2013), 249). Furthermore, given the involvement of above and below ground biomass in a number of mitigation measures, ecosystem-based adaptation could be extended to ecosystem-based mitigation (e.g. urban trees, Davis *et al*., 2011).

To remove barriers and further make the goal of successful win-win strategies achievable, a number of authors highlight the need for international and national governance of adaptation (e.g. Tol, 2005). The White Paper further emphasises the need for higher-level governance to increase the level of integration across different sectors and governance levels, and the effectiveness of actions. This suggests that, although as a result of regional variations of the impact of climate change, strategies will be conducted at national, regional and local levels; they can be improved and integrated using a coordinated approach by the EU. Sectors which require such an approach include agriculture, water and biodiversity, as they are inherently integrated at an EU level by single markets and common policies, but also can be transnational in nature.

More recently, there has been a shift in emphasis to increasing climate resilience and improving coordination, as seen in the 2013 EU Adaptation Strategy (SWD 2013, 216), which illustrates progress made since the White Paper. EU member states are being encouraged to adapt NASs, such as those discussed earlier, to increase the uptake of win-win no-regret strategies, such as sustainable water management. The need for existing EU policies, such as CAP, is also highlighted in addition to improving knowledge through strategies, such as the Marine Knowledge 2020 Strategy (COM (2012) 473 final). The strategy illustrates the development of tools at an EU level, such as Climate-ADAPT (SWD(2013), 134) to aid decision-making for adaptation within Europe and highlights the role of the EU to fill knowledge gaps in adaptation and ensure that action is being taken

The number of interactions between policy sectors is growing (see Figure 3), and such a high level of interaction highlights the need for an integrated approach which can work over multiple domains and scales to communicate on the border of science and policy. Integrated climate governance (ICG) is identified in the current literature as one such way forward for mitigation and adaptation (Tàbara, 2011). Such international governance would consider multiple domains, scales and governance levels. It would most importantly demand institutional innovation to facilitate (1) policy instruments and measures, (2) improvements to climate resilience, communication and learning, and (3) tools and methods for climate assessment, whilst (4) linking global level processes with those taken at local and regional levels. ICG would act to provide guidance for scientists, policy-makers and stakeholders, and analyse current practices (Tàbara, 2011).

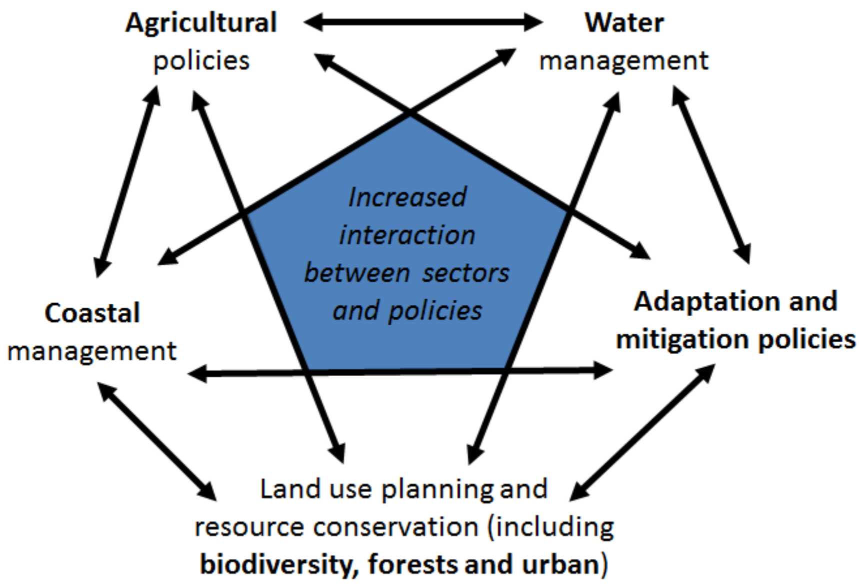

Figure 3: The cross-sectoral nature of policies and management. Adapted from Tàbara (2011).

Both the concepts of EU-level coordination and IGC provide novel approaches to the management of adaptation and mitigation strategies. Findings from this review have highlighted the number of complex cross-sectoral interactions strategies can have, and the plethora of actors and governance levels involved. Although much has been done, and there are clear examples of win-win strategies, current practices and policies could be much improved by high-level integrated governance of climate change mitigation and adaptation, and adaptation issues.

17. Conclusions

Despite the high level calls for action on adaptation and mitigation and for their mainstreaming into policy, there is a lack of information on some measures. Even those for which there is information on their implementation, there is often a lack of evidence on their effectiveness and wider impacts. This is partly due to little long-term monitoring of the strategies (Adger *et al*., 2005) and to the time taken for the success of some measures to become evident. Also, in the case of biodiversity, for example, there is not always a clear distinction between good management practice and what is needed specifically for climate change, as resilient ecosystems are more likely to be able to adapt autonomously and require less intervention (e.g. Tompkins and Adger, 2004; COM (2013), 249).

As has been clearly shown, many adaptation and mitigation measures interact with each other, although the examples tended to demonstrate how adaptation could contribute to mitigation, rather than how mitigation can contribute to adaptation. It would be valuable to explore this further through a more thorough review of the mitigation literature. This review also showed how the sectoral adaptation and mitigation measures interact within the intended sector and with other sectors. It identified that the cross-sectoral interactions may be beneficial to both sectors (synergistic) or be positive on the implementing sector and have a negative effect on the impacted sector (antagonistic). The largest category of synergies identified involved those between adaptation and mitigation within a sector. Often these synergies and antagonisms were not explicit in the literature and if more successful adaptation and mitigation is to be undertaken these need to be stated explicitly and the benefits of measures quantified, in order that greater effectiveness can be achieved or trade-offs dealt with in the case of antagonistic interactions (see Stoorvogel *et al*., 2004; Jarvis *et al*., 2011). This would require greater cross-sectoral working and integration across relevant policies at all levels of governance.

In order to put this into practice, it might be good to begin with biodiversity and water as there were a high number of synergies associated with these sectors. Also, adaptation and mitigation measures both within these sectors and impacting on these sectors were shown to have a cross-cutting nature. To some extent the importance of biodiversity is already being recognised through the development of policies relating to green infrastructure and the promotion of ecosystem-based adaptation and, given the involvement of above and below ground biomass in a number of mitigation measures, this could be extended to ecosystem-based mitigation. Whilst for water, the Water Framework Directive provides some opportunity for integration of adaptation across sectors, but there is also room for other sectors undertaking water-based adaptation measures to connect directly with the water sector.

The review found that both adaptation and mitigation measures tended to be implemented at local to regional scales, although there are those which form part of national planning (e.g. some coastal management strategies, such as the UKs intertidal agri-environmental scheme (Parrott and Burningham, 2008) or benefit from international cooperation. There is an opportunity, if not indeed an urgent need, for them to be more integrated at the local to regional scale in order that the identified synergies can be realised and the antagonisms avoided. This integration could be extended to cover the transnational component of some sectors or measures, for example, ecological networks and river basin management. While the implementation of strategies may be at the local to regional scale, many regulations and policies governing practice stem from European level policies (e.g. Wolfe *et al*., 2008; Aste *et al.*, 2010). Thus there is a need to ensure that they can be translated into relevant policy and practice at the more local scale and for local experiences to feedback, such that there is reflexive policy formulation (Urwin and Jordan, 2008), much as is advocated for adaptive management. This interaction is necessary to ensure appropriate policy formulation and to address the complexity surrounding issues of governance, institutions and actors.

In addition to achieving adaptation and/or mitigation, many of the measures examined had other environmental or social benefits, for example, improvements in air quality (Tallis *et al.*, 2011) and new recreational opportunities (Luisetti *et al*., 2011; Thiere *et al.*, 2011). Some of these are explicitly part of the adaptation measures, such as schemes designed to reduce human heat stress, while those pertaining to aesthetics, recreation and tourism may not be viewed as an explicit part of the design of the adaptation or mitigation measure. They may, however, indicate additional opportunities and benefits and represent not insignificant economic returns on investment. The evidence on cost-effectiveness was most forthcoming for the coastal and urban sectors, but nearly all those measures for which there were figures showed positive economic benefits, although for some with long lead times or high development costs, the costs will take a long while to recover (e.g. the Oosterschelde storm-surge barrier, Smits *et al.,* 2006).

In conclusion, the majority of the adaptation and mitigation measures examined had synergies with other sectors, with fewer antagonisms being identified, albeit many of these were not explicitly stated. For these synergies to be realised it will require cross-sectoral working which presents challenges, as it will in turn require interactions across governance levels, as well as engagement with multiple stakeholders. It will, however, provide opportunities for more efficient, often cost-effective actions to be undertaken. This will require appropriate metrics for the consistent assessment of which measures are the most effective, but currently ecosystem-based adaptation or mitigation and green infrastructure seem promising as they involve a high number of synergies and benefit multiple sectors. This review has shown that, in addition to adaptation and mitigation being necessary responses to climate change, the cross-sectoral integration of adaptation and mitigation measures can provide resource-efficient responses, with other potential environmental and social benefits. There are challenges to this cross-sectoral integration, but some mainstreaming is under way, and there are clear examples of win-win strategies. Current practices and policies could be much improved by high-level integrated governance of climate change mitigation and adaptation to ensure that Europe is responding effectively to the challenges and opportunities of climate change adaptation and mitigation.

18. References

Adams, R.M. and Cho, S.H., (1998). Agriculture and endangered species: An analysis of trade-offs in the Klamath Basin, Oregon. Water Resources Research, 34(10): 2741-2749.

Adams, R.M., McCarl, B.A., & Mearns, L.O. (2003). The effects of spatial scale of climate scenarios on economic assessments: An example from US agriculture. Climatic Change, 60(1-2): 131-148.

Adger, W.N., Arnell, N.W., Tompkins, E.L. (2005). Successful adaptation to climate change across scales. Global Environmental Change, 15: 77-86.

Adhikari, R. S., Buzzetti, M., & Magelli, S. (2011). Solar photovoltaic and thermal systems for electricity generation, space heating and domestic hot water in a residential building. In Clean Electrical Power (ICCEP), 2011 International Conference on (pp. 461-465). IEEE.

Agobia, C.A. (1999). Enhancing sustainable livelihoods in drought prone areas of Mudzi (Makaha Ward) and Gwanda (Gwanda Ward 19). Building on Adaptive Strategies. IISD, Community Drought Mitigation Project, Final Report Project Number 050/19284, 34 pp.

Agrawala, S. & van Aalst, M. (2005). Bridging the gap between climate change and development. *Bridge Over Troubled Waters: Linking Climate Change and Development*, OECD, Paris, pp. 133-146.

Agrawala, S., Kramer, A., Prudent-Richard, G., Sainsbury, M., & Schreitter, V. (2012). Incorporating climate change impacts and adaptation in environmental impact assessments: opportunities and challenges. Climate and Development, 4(1): 26-39.

Airoldi, L., Abbiati, M., Beck, M.W., Hawkins, S.J., Jonsson, P.R., Martin, D., Moschella, P.S., Sundelöf, A., Thompson, R.C., Åberg, P. (2005). An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. Coastal Engineering 52(10-11): 1073-1087.

Akbari, H., & Matthews, H. D. (2012). Global cooling updates: Reflective roofs and pavements. Energy and Buildings, 55: 2-6.

Akbari, H., Pomerantz, M., & Taha, H. (2001). Cool surfaces and shade trees to reduce energy use and improve air quality in urban areas. Solar energy, 70(3): 295-310.

Akiyama, H., Yagi, K., Yan, X.Y. (2005). Direct N2O emissions from rice paddy fields: Summary of available data. Global Biogeochemical Cycles 19(1): GB1005.

Alexandri, E., & Jones, P. (2008). Temperature decreases in an urban canyon due to green walls and green roofs in diverse climates. Building and Environment, 43(4): 480-493.

Alexandrov, V., Eitzinger, J., Cajic, V., & Oberforster, M. (2002). Potential impact of climate change on selected agricultural crops in north-eastern Austria. Global Change Biology, 8(4): 372-389.

Allen, S.R., & Hammond, G.P. (2010). Thermodynamic and carbon analyses of micro-generators for UK households. Energy, 35(5): 2223-2234.

Álvaro-Fuentes, J., López, M.V., Cantero-Martinez, C., & Arrúe, J.L. (2008). Tillage effects on soil organic carbon fractions in Mediterranean dryland agroecosystems. Soil Science Society of America Journal, 72(2): 541-547.

Álvaro-Fuentes, J., Morell, F.J., Plaza-Bonilla, D., Arrúe, J.L., & Cantero-Martínez, C. (2012). Modelling tillage and nitrogen fertilization effects on soil organic carbon dynamics. Soil and Tillage Research, 120: 32-39.

Ancell, S. & Thompson-Fawcett, M. (2008). The Social Sustainability of Medium Density Housing: A Conceptual Model and Christchurch Case Study. Housing Studies 23(3): 423-442.

Andersen, C.T., Foster, I.D.L., & Pratt, C.J. (1999). The role of urban surfaces (permeable pavements) in regulating drainage and evaporation: development of a laboratory simulation experiment. Hydrological Processes, 13(4): 597-609.

Anderson, M., Lambrinos, J., & Schroll, E. (2010). The potential value of mosses for stormwater management in urban environments. Urban Ecosystems, 13(3): 319-332.

Andrews, J. E., Burgess, D., Cave, R.R., Coombes, E.G., Jickells, T.D., Parkes, D.J., Turner, R.K., (2006). Biogeochemical value of managed realignment, Humber estuary, UK. Science of the Total Environment, 371(1-3): 19-30.

Andrews, J. E., Samways, G., Shimmield, G.B., (2008). Historical storage budgets of organic carbon, nutrient and contaminant elements in saltmarsh sediments: Biogeochemical context for managed realignment, Humber Estuary, UK. Science of the Total Environment, 405(1-3): 1-13.

Anon (1996). Solar energy dwellings in the Netherlands. IEEE Power Engineering Review, 2pp.

Antle, J. and Ogle, S. (2012). Influence of soil C, N2O and fuel use on GHG mitigation with no-till adoption. Climatic Change, 111(3): 609-625.

Araújo, M.B., Alagador, D., Cabeza, M., Nogués-Bravo, D., & Thuiller, W. (2011). Climate change threatens European conservation areas. Ecology letters, 14(5): 484-492.

Araújo, M.B., Cabeza, M., Thuiller, W., Hannah, L., & Williams, P.H. (2004). Would climate change drive species out of reserves? An assessment of existing reserve-selection methods. Global Change Biology, 10(9): 1618-1626.

Araus, J.L., Slafer, G.A., Reynolds, M.P., & Royo, C. (2002). Plant breeding and drought in C3 cereals: what should we breed for? Annals of Botany, 89(7): 925-940.

Armson, D., Stringer, P., & Ennos, A.R. (2012). The effect of tree shade and grass on surface and globe temperatures in an urban area. Urban Forestry & Urban Greening, 11(3): 245-255.

Arnell, N.W. & Delaney, E. K. (2006). Adapting to climate change: Public water supply in England and Wales. Climatic Change, 78: 227-255.

Artmann, N., Manz, H., & Heiselberg, P. (2008). Parameter study on performance of building cooling by night-time ventilation. Renewable Energy, 33(12): 2589-2598.

Asadian, Y., & Weiler, M. (2009). A new approach in measuring rainfall interception by urban trees in coastal British Columbia. Water quality research journal of Canada, 44(1): 16.

Aste, N., Adhikari, R. S., & Buzzetti, M. (2010). Beyond the EPBD: the low energy residential settlement Borgo Solare. Applied Energy, 87(2): 629-642.

Åström, M., Dynesius, M., Hylander, K., & Nilsson, C. (2007). Slope aspect modifies community responses to clear-cutting in boreal forests. Ecology, 88(3):749-758.

Bahaj, A.S., & James, P.A.B. (2007). Urban energy generation: the added value of photovoltaics in social housing. Renewable and Sustainable Energy Reviews, 11(9): 2121-2136.

Bakker, J.P., Esselink, P., Dijkema, K.S., van Duin, W.E., de Jong, D.J., (2002). Restoration of salt marshes in the Netherlands. Hydrobiologia, 478(1): 29-51.

Ball, B.C., Scott, A., & Parker, J. P. (1999). Field N2O, CO2 and CH4 fluxes in relation to tillage, compaction and soil quality in Scotland. Soil and Tillage Research, 53(1): 29-39.

Balletto, J.H., Heimbuch, M.V., Mahoney, H.J., (2005). Delaware Bay salt marsh restoration: Mitigation for a power plant cooling water system in New Jersey, USA. Ecological Engineering, 25(3): 204-213.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D. & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters, 9: 1146–1156.

Barkowski, J.W., Kolditz, K., Brumsack, H., Freund, H. (2009). The impact of tidal inundation on salt marsh vegetation after de-embankment on Langeoog Island, Germany - Six years time series of permanent plots. Journal of Coastal Conservation, 13(4): 185-206.

Barlow, C., & Martin, P.S. (2007). Wild Earth Forum [Fall/Winter] p 52-56.

Bartlett, K. B., Harriss, R.C., Sebacher, D.I. (1985). Methane flux from coastal salt marshes. Journal of Geophysical Research, 90(D3): 5710-5720.

Bates, A.J., Mackay, R., Greswell, R.B., & Sadler, J.P. (2009). SWITCH in Birmingham, UK: experimental investigation of the ecological and hydrological performance of extensive green roofs. Reviews in Environmental Science and Bio/Technology, 8(4): 295-300.

Battle, G., Yuen, C.H.N., Zanchetta, M., & D'Cruz, P. (2006). Energy efficiency in new museum build: THEpUBLIC. International Journal of Sustainable Energy, 25(3-4): 185-198.

Baumann, N. (2006). Ground-nesting birds on green roofs in Switzerland: Preliminary observations. Urban Habitats, 4(1): 37-50.

Beeftink, W. G. (1975). The Ecological Significance of Embankment and Drainage with Respect to the Vegetation of the South-West Netherlands. Journal of Ecology, 63(2): 423-458.

Belder, P., Spiertz, J.H.J., Bouman, B.A.M., Lu, G., & Tuong, T.P. (2005). Nitrogen economy and water productivity of lowland rice under water-saving irrigation. Field Crops Research 93(2):169-185.

Bell, M. J., Wall, E., Simm, G., & Russell, G. (2011). Effects of genetic line and feeding system on methane emissions from dairy systems. Animal Feed Science and Technology, 166: 699-707.

Bellocq, M.I., Smith, S.M., Doka, M.E. (2001). Short-term effects of harvest technique and mechanical site preparation on arthropod communities in jack pine plantations. Journal of Insect Conservation, 5(3): 187-196.

Benedict, M.A., & McMahon, E.T. (2002). Green infrastructure: smart conservation for the 21st century. Renewable Resources Journal, 20(3): 12-17.

Benemann, J., & Chehab, O. (1996). Pilkington solar international for PV applications in building. Renewable Energy, 8(1): 452-457.

Bennetzen, E. H., Smith, P., Soussana, J. F., & Porter, J. R. (2012). Identity-based estimation of greenhouse gas emissions from crop production: Case study from Denmark. European Journal of Agronomy, 41: 66-72.

Berkooz, C.B. (2011). Green Infrastructure Storms Ahead. Planning, 77(3): 19-24.

Berman, A. (2005). Estimates of heat stress relief needs for Holstein dairy cows. Journal of Animal Science, 83(6): 1377-1384.

Bernhardt, K.-G. and Koch, M. (2003). Restoration of a salt marsh system: temporal change of plant species diversity and composition. Basic and Applied Ecology, 4(5): 441-451.

Bernier, P., & Schoene, D. (2009). Adapting forests and their management to climate change: an overview. Unasylva 231/232, 60: 7pp.

Berry, P. (2009b). Costing Adaptation for Natural Ecosystems in Parry, M., Arnell, N., Berry, P., Dodman, D., Fankhauser, S., Hope, C., Kovats, S., Nicholls, N., Satterthwaite, D., Tiffin, R. and Wheeler, T. (2009). Assessing the costs of adaptation to climate change: A critique of the UNFCCC estimates. IIED, London, pp90-991.

Berry, P., Burch, S., Sanders, M. (2010). EMBEDS: Embedding Biodiversity Adaptation Principles. Final report prepared for United Kingdom Department of Environment, Food and Rural Affairs. UK Defra, London, UK.

Berry, P., Paterson, J., Cabeza, M., Dubuis, A., Guisan, A., Jäättelä, L., Kühn, I., Midgley, G., Musche, M., Piper, J., Wilson, E. (2008a). MACIS - Minimisation of and Adaptation to Climate Change Impacts on Biodiversity. Deliverables 2.2 and 2.3: Meta-analysis of adaptation and mitigation measures across the EU25 and their impacts and recommendations how negative impacts can be avoided, Oxford, http://www.macis-project.net/pub.html Accessed: July 2010.

Berry, P., Paterson, J., Cabeza, M., Dubuis, A., Guisan, A., Jäättelä, L., Kühn, I., Midgley, G., Musche, M., Piper, J., & Wilson, E. (2008b). Minimisation of and Adaptation to Climate change: Impacts on biodiversity.

Berry, P.M. (Ed.) (2009a). Biodiversity in the Balance – Mitigation and Adaptation Conflicts and Synergies. Pensoft Publishing, Sofia, Bulgaria.

Berry, P.M. and Paterson, J.S. (2010) Climate Change Mitigation and Adaptation Measures and Biodiversity. In Atlas of Biodiversity Risk J. Settele, R. Grabaum, V. Grobelnick, V. Hammen, S. Klotz, L. Penev, I. Kühn (eds.). Pensoft, Sofia, Moscow.

Bescansa, P., Imaz, M. J., Virto, I., Enrique, A., & Hoogmoed, W. B. (2006). Soil water retention as affected by tillage and residue management in semiarid Spain. Soil and Tillage Research, 87(1): 19-27.

Beunen, R. (2006). European nature conservation legislation and spatial planning: For better or for worse? Journal of environmental planning and management, 49(4): 605-619.

Bidegain, P.M., García, P.F., and Methol, R. (2001). Long-term effect of tillage intensity for Eucalyptus grandis planting on some soil physical properties in an Uruguayan Alfisol. In 3rd International Conference on Land Degradation and Meeting of IUSS Subcommission C-Soil and Water Conservation.

Biesbroek, G.R., Swart, R.J. and van der Knaap, W.G. (2009). The mitigation–adaptation dichotomy and the role of spatial planning. Habitat international, 33(3): 230-237.

Biesbroek, G.R., Swart, R.J., Carter, T.R., Cowan, C., Henrichs, T., Mela, H., Morecroft, M.D., Rey, D., (2010). Europe adapts to climate change: comparing national adaptation strategies. Global Environmental Change, 20(3): 440-450

Bigham, R. (2011). The Little Details. Pollution Engineering, 43(4): 7-7.

BioRegional (2002). Beddington Zero Energy Development: case study report. Housing Corporation, 16pp.

Birrell, B., O'Connor, K., Rapson, V., & Healy, E. (2005). Melbourne 2030: Planning rhetoric versus urban reality, Monash University ePress, 126pp.

Bishop, M.J., Peterson, C.H., Summerson, H.C., Lenihan, H.S., Grabowski, J.H. (2006). Deposition and long-shore transport of dredge spoils to nourish beaches: Impacts on benthic infauna of an ebb-tidal delta. Journal of Coastal Research, 22(3): 530-546.

Biswas, B., Scott, P.T., & Gresshoff, P.M. (2011). Tree legumes as feedstock for sustainable biofuel production: Opportunities and challenges. Journal of plant physiology, 168(16): 1877-1884.

Blackwell, M.S.A., Hogan, D.V., Maltby, E., (2004). The short-term impact of managed realignment on soil environmental variables and hydrology. Estuarine, Coastal and Shelf Science, 59(4): 687-701.

Blackwell, M.S.A., Yamulki, S., Bol, R. (2010). Nitrous oxide production and denitrification rates in estuarine intertidal saltmarsh and managed realignment zones. Estuarine, Coastal and Shelf Science, 87(4): 591-600.

Blanco-Canqui, H. and Lal, R. (2008). No-tillage And Soil-profile Carbon Sequestration: An On-farm Assessment. Soil Sci. Soc. Am. J. 72(3): 693-701.

Blockley, D.J. & Chapman, M.G. (2008). Exposure of seawalls to waves within an urban estuary: effects on intertidal assemblages. Austral Ecology 33(2): 168-183.

BMU (2008). German strategy for adaptation to climate change. German Federal Cabinet, 73pp. Available: http://www.bmu.de/files/english/pdf/application/pdf/das_gesamt_en_bf.pdf Accessed: June 2013.

Bøckman, O. C. (1997). Fertilizers and biological nitrogen fixation as sources of plant nutrients: perspectives for future agriculture. Plant and soil, 194(1-2): 11-14.

Bohlool, B. B., Ladha, J. K., Garrity, D. P., & George, T. (1992). Biological nitrogen fixation for sustainable agriculture: A perspective. In Biological nitrogen fixation for sustainable agriculture (pp. 1-11). Springer Netherlands.

Bonesmo, H., Skjelvåg, A.O., Henry Janzen, H., Klakegg, O., & Tveito, O.E. (2012). Greenhouse gas emission intensities and economic efficiency in crop production: A systems analysis of 95 farms. Agricultural Systems, 110: 142-151.

Borin, M., Menini, C., & Sartori, L. (1997). Effects of tillage systems on energy and carbon balance in north-eastern Italy. Soil and Tillage Research, 40(3): 209-226.

Bosello, F., Carraro, C., & de Cian, E. (2013). Adaptation can help mitigation: an integrated approach to post-2012 climate policy. Environment and Development Economics 18(3): 270-290.

Boven, K. & Morohashi, J. (Eds.) (2002). Best Practices using Indigenous Knowledge. Nuffic, The Hague, The Netherlands, and UNESCO/MOST, Paris, France. Available at: http://www.unesco.org/most/Bpikpub2.pdf Accessed: July 2012.

Bowler, D.E., Buyung-Ali, L., Knight, T.M., & Pullin, A.S. (2010). Urban greening to cool towns and cities: A systematic review of the empirical evidence. Landscape and urban planning, 97(3): 147-155.

Bozek, C.M., & Burdick, D.M. (2005). Impacts of Seawalls on Saltmarsh Plant Communities in the Great Bay Estuary, New Hampshire USA. Wetlands Ecology and Management, 13(5): 553-568.

BRANCH, 2007, Planning for biodiversity in a changing climate – BRANCH project Final Report', Natural England, UK, 32pp.

Brehmer, B., Struik, P.C., & Sanders, J. (2008). Using an energetic and exergetic life cycle analysis to assess the best applications of legumes within a biobased economy. Biomass and Bioenergy, 32(12): 1175-1186.

Brenneisen, S. (2006). Space for urban wildlife: designing green roofs as habitats in Switzerland. Urban Habitats, 4(1): 27-36.

Brink, J.C., Hordijk, L., Van Ierland, E.C., & Kroeze, C. (2000). Cost-effective N2O, CH4 and NH3 abatement in European agriculture: interrelations between global warming and acidification policies. In Expert Workshop on Assessing the Ancillary Benefits and Costs of Greenhouse Gas Mitigation Strategies (pp. 27-29).

Broad, W., & Barbarito, B. (2004). Examination of site selection methodologies for the retrofit of SUDS. In N.J.Horan, Ed., Proceedings of the Second National Conference of the Chartered Institution of Water and Environmental Management, Wakefield, England, 319pp..

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity and Conservation, 17(5): 925-951.

Bruns, E. (2007). Bewertungs-und Bilanzierungsmethoden in der Eingriffsregelung. Analyse und Systematisierung von Vorgehensweisen des Bundes und der Länder. PhD Institut für Landschaftsarchitektur und Umweltplannung, TU Berlin, Germany.

BSPB (2010). Top breeder calls for UK wheat probe. British Society of Plant Breeders. Plant breeding Matters, 8: 2-3.

Buijs, A.E. (2009). Public support for river restoration. A mixed-method study into local residents' support for and framing of river management and ecological restoration in the Dutch floodplains. Journal of Environmental Management, 90: 2680-2689.

Bulleri, F., & Chapman, M.G. (2004). Intertidal assemblages on artificial and natural habitats in marinas on the north-west coast of Italy. Marine Biology, 145(2): 381-391.

Bulleri, F., & Chapman, M.G., (2010). The introduction of coastal infrastructure as a driver of change in marine environments. Journal of Applied Ecology, 47(1): 26-35.

Bunce, S. (2004). The emergence of "smart growth" intensification in Toronto: environment and economy in the new official plan. Local Environment, 9(2): 177-191.

Bunce, S. (2009). Developing sustainability: sustainability policy and gentrification on Toronto's waterfront. Local Environment, 14(7): 651-667.

Burch, S. and Berry, P.M. (2013). Embedding climate change adaptation in biodiversity conservation: A case study of England. Environmental Science and Policy, 23pp, submitted.

Burch, S. (2010a). In pursuit of resilient, low-carbon communities: An examination of barriers to action in three Canadian cities. Energy Policy, 38(2): 7575-7585.

Burch, S. (2010b). Transforming barriers into enablers of action on climate change: Insights from three case studies in British Columbia, Canada. Global Environmental Change, 20(2): 287-297.

Burkett, V., & Kusler, J. (2000). Climate change: potential impacts and interactions in wetlands of the United States. Journal of the American Water Resources Association, 36(2): 313-320.

Burkett, V., & Kusler, J. (2000). Climate change: potential impacts and interactions in wetlands of the United States. Journal of the American Water Resources Association, 36(2): 313-320.

Butler D, Davies. (2004) Urban Drainage, E & FN Spon.

Campbell, A., & Bradfield, G.E. (1988). Short-term vegetation change after dyke breaching at the Kokish marsh, northeastern Vancouver Island. Northwest Science, 62(1): 28-35.

Cantero-Martínez, C., Angás, P., & Lampurlanés, J. (2007). Long-term yield and water use efficiency under various tillage systems in Mediterranean rainfed conditions. Annals of Applied Biology, 150(3): 293-305.

Cao, S., Chen, L., Feng, Q., & Liu, Z. (2007). Soft-riser bench terrace design for the hilly loess region of Shaanxi Province, China. Landscape and urban planning, 80(1): 184-191.

Carbonell-Bojollo, R., González-Sánchez, E.J., Veróz-González, O., Ordóñez-Fernández, R. (2011). Soil management systems and short term CO2 emissions in a clayey soil in southern Spain. Science of the Total Environment, 409(15): 2929-2935.

Carlton, R.R., West, J.S., Smith, P., & Fitt, B.D. (2012). A comparison of GHG emissions from UK field crop production under selected arable systems with reference to disease control. European Journal of Plant Pathology, 133(1): 333-351.

Carneiro, G. (2007). The parallel evolution of ocean and coastal management policies in Portugal. Marine Policy, 31(4): 421–433

Carrasco-Letelier, L., Eguren, G., Castiñeira, C., Parra, O., & Panario, D. (2004). Preliminary study of prairies forested with Eucalyptus sp. at the northwestern Uruguayan soils. Environmental Pollution, 127(1): 49-55.

Carroll, M.J., Anderson, B.J., Brereton, T.M., Knight, S.J., Kudrna, O., & Thomas, C.D. (2009). Climate change and translocations: the potential to re-establish two regionally-extinct butterfly species in Britain. Biological Conservation, 142(10): 2114-2121.

Carvalho, S.B., Brito, J.C., Crespo, E.G., Watts, M.E., & Possingham, H.P. (2011). Conservation planning under climate change: Toward accounting for uncertainty in predicted species distributions to increase confidence in conservation investments in space and time. Biological Conservation, 144(7): 2020-2030.

Casal-Campos, A., Jefferies, C., & Momparler, S.P. (2012). Selecting SUDS in the Valencia Region of Spain. Water Practice and Technology, 7(1): 9pp.

CAST (1992). Council of Agriculture and Science Technology Preparing U.S. agriculture for global climate change. Council for Agriculture and Technology, Report 119, Washington,
Ceccarelli, S., Grando, S. & Baum, M. (2007). Participatory plant breeding in water-limited environments. Experimental Agriculture, 43: 411–435.

Ceccato, L., Giannini, V. & Giupponi, C. (2011). Participatory assessment of adaptation strategies to flood risk in the Upper Brahmaputra and Danube river basins. Environmental Science & Policy, 14: 1163-1174.

Challinor, A.J., Smith, M.S., & Thornton, P. (2012). Use of agro-climate ensembles for quantifying uncertainty and informing adaptation. Agricultural and Forest Meteorology, 170: 2-7.

Chance, T. (2009). Towards sustainable residential communities; the Beddington Zero Energy Development (BedZED) and beyond. Environment and Urbanization, 21(2): 527-544.

Chang, Y.H., Scrimshaw, M.D., MacLeod, C.L., Lester, J.N., (2001). Flood defence in the Blackwater Estuary, Essex, UK: the impact of sedimentological and geochemical changes on salt marsh development in the Tollesbury managed realignment site. Marine Pollution Bulletin, 42(6): 469-480.

Chapin III, F.S., Danell, K., Elmqvist, T., Folke, C., & Fresco, N. (2007). Managing climate change impacts to enhance the resilience and sustainability of Fennoscandian forests. AMBIO: A Journal of the Human Environment, 36(7): 528-533.

Chapman, M.G. & Bulleri, F. (2003). Intertidal seawalls-new features of landscape in intertidal environments. Landscape and Urban Planning, 62(3): 159-172.

Chapman, M.G. (2003). Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. Marine Ecology Progress Series, 264: 21-29.

Charlton, M.B. & Arnell, N.W. (2011). Adapting to climate change impacts on water resources in England—an assessment of draft water resources management plans. Global Environmental Change, 21: 238-248.

Chatskikh, D., Olesen, J.E., Hansen, E.M., Elsgaard, L., & Petersen, B.M. (2008). Effects of reduced tillage on net greenhouse gas fluxes from loamy sand soil under winter crops in Denmark. Agriculture, Ecosystems & Environment, 128(1): 117-126.

Chazdon R. L. (2009). Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands. Science, 320(5882): 1458-1460.

Chen, C., Pan, X.B., Zhang, L.Z., Pang, Y.M. (2011). Impact of climate change on cotton production and water consumption in Shiyang River Basin. Transactions of the CSAE 27(1): 57-65. (in Chinese with English abstract)

Chen, L., Zuo, T., Rasaily, R.G. (2010). Farmer's adaptation to climate risk in the context of China. Agriculture and Agricultural Science Procedia, 1: 116-125.

Cherry, K., Mooney, S.J., Ramsden, S., & Shepherd, M.A. (2012). Using field and farm nitrogen budgets to assess the effectiveness of actions mitigating N loss to water. Agriculture, Ecosystems & Environment, 147: 82-88.

Choi, Y. & Wang, Y. (2004). Dynamics of carbon sequestration in a coastal wetland using radiocarbon measurements. Global Biogeochemical Cycles, 18(4): GB4016.

Choi, Y., Wang, Y., Hsieh, Y.-P., Robinson, L., (2001). Vegetation Succession and Carbon Sequestration in a Coastal Wetland in Northwest Florida: Evidence From Carbon Isotopes. Global Biogeochemical Cycles, 15(2): 311-319.

Clar, C., Prutsch, A., & Steurer, R. (2013). Barriers and guidelines for public policies on climate change adaptation: A missed opportunity of scientific knowledge-brokerage. Natural Resources Forum, 37(1): 1-18.

Clark, C., Adriaens, P., & Talbot, F.B. (2008). Green roof valuation: a probabilistic economic analysis of environmental benefits. Environmental science & technology, 42(6): 2155-2161.

Clark, N. A. (2006). Tidal Barrages and Birds. Ibis 148: 152-157.

Coffman, R.R., & Davis, G. (2005). Insect and avian fauna presence on the Ford assembly plant ecoroof. In: Third annual greening rooftops for sustainable communities conference, awards and trade show (pp. 4-6).

Colclough, S., Griffiths, P., & Gschwander, S. (2009). Thermal Energy Storage and the Passive House Standard: How PCM incorporated into wallboard can aid thermal comfort. PLEA2009 - The 26th Conference on Passive and Low Energy Architecture.

COM (2009). 147 final, EU White Paper. Adapting to climate change: towards a European framework for action. Commission of the European Communities, Brussels, 17pp.

COM (2012). 273 final, Green Paper. Marine Knowledge 2020: from seabed mapping to ocean forecasting. SWD (2012) – 250 final, European Commission, Brussels, 19pp.

COM (2013). 249 final. Technical information on Green Infrastructure (GI). Enhancing Europe's natural capital - SWD (2013) 155 final. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. European Commission, Brussels, 24pp.

Connor, J. D., Schwabe, K., King, D., & Knapp, K. (2012). Irrigated agriculture and climate change: The influence of water supply variability and salinity on adaptation. Ecological Economics, 77: 149-157.

Connor, R.F., Chmura, G.L., Beecher, B., (2001). Carbon accumulation in Bay of Fundy salt marshes: Implications for restoration of reclaimed marshes. Global Biogeochemical Cycles, 15(4): 943-954.

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P., & Beaudoin, N. (2010). Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agriculture, ecosystems & environment, 135(4): 268-278.

Convention on Biological Diversity. 2005. Report of the meeting of the Ad Hoc Technical Expert Group on Biodiversity and Adaptation to Climate 17 Change. NEP/CBD/SBSTTA/11/INF/5.

Cooper, C.M. & Knight, S.S. (1990). Nutrient trapping efficiency of a small detention reservoir. Agricultural Water Management, 18: 149-158.

Cooper, P., Rao, K.P.C., Singh, P., Dimes, J., Traore, P.S., Dixit, P., Twomlow, S.J. (2009). Farming with current and future climate risk: advancing a 'Hypothesis of Hope' for rainfed agriculture in the semi-arid tropics. Journal of SAT Agricultural Research, 7: 1-19.

Corgnati, S.P., Kindins, A., & Perino, M. (2007). Phase change materials with ceiling ventilation system to reduce summer cooling loads. The 6th International Conference on Indoor Air Quality, Ventilation & Energy Conservation in Buildings IAQVEC 2007, Sendai, Japan

Coutts, C. (2010). Green Infrastructure and Public Health in the Florida Communities Trust Public Land Acquisition Program. Planning Practice & Research, 25(4): 439-459.

Cowling, D. W. (1982). Biological Nitrogen Fixation and Grassland Production in the United Kingdom. Philosophical Transactions of the Royal Society of London. B, Biological Sciences 296(1082): 397-404.

Crabbé, P., Robin, M., (2006). Institutional adaptation of water resource infrastructures to climate change in eastern Ontario. Climatic Change, 78: 103-133.

Crowther, A.E. (2007). The restoration of intertidal habitats for non-breeding waterbirds through breached managed realignment Stirling. Doctoral dissertation, University of Stirling.

Cuculeanu, V., Marica, A., and Simota, C. (1999). Climate Change Impact on Agricultural Crops and Adaptation Options in Romania. Climate Research, 12: 153–160.

Cullen, B. R., & Eckard, R. J. (2011). Impacts of future climate scenarios on the balance between productivity and total greenhouse gas emissions from pasture based dairy systems in south-eastern Australia. Animal Feed Science and Technology, 166: 721-735.

Da Silva, J., F., Duck, R., W. (2001). Historical changes of bottom topography and tidal amplitude in the Ria de Aveiro, Portugal - trends for future evolution. Climate Research, 18: 17-24.

Daccache, A., Keay, C., Jones, R.J.A., Weatherhead, E.K., Stalham, M.A., & Knox, J.W. (2012). Climate change and land suitability for potato production in England and Wales: impacts and adaptation. Journal of Agricultural Science, 150(2): 161.

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., Ayres, ,M.P., Flannigan, M.D., Hanson, P.J., Irland, L.C., Lugo, A.E., Peterson, C.J., Simberloff, D., Swanson, F.J., Stocks, B.J., Wotton, B.M. (2001). Climate change and forest disturbances. BioScience, 51:723–734

Dalgaard, T., Olesen, J.E., Petersen, S.O., Petersen, B.M., Jørgensen, U., Kristensen, T., Hutchings, N.J., Gyldenkærne, S., & Hermansen, J.E. (2011). Developments in greenhouse gas emissions and net energy use in Danish agriculture – How to achieve substantial CO2 reductions? Environmental Pollution, 159(11): 3193-3203.

Dang, X.L., Huang, Y., Yu, N., Zhang, Y.L (2006) Status quo and existing problems of water saving irrigation in protected field of Liaoning province. Water Saving Irrigation, 5: 57-59.

Danso, S.K.A., Bowen, G.D., & Sanginga, N. (1992). Biological nitrogen fixation in trees in agro-ecosystems. Plant and Soil, 141(1-2): 177-196.

Darnell, J. T. & Heilman, D.J. (2007). Fringe Benefits. Civil Engineering, 77(5): 50-55.

Das Gupta, A. & Babel, M.S. (2005). Challenges for sustainable management of groundwater in Bangkok, Thailand. International Journal of Water Resources Development, 21(3): 453-463.

Davidson, I., & Simkanin, C. (2008). Skeptical of assisted colonization. Science (New York, NY), 322(5904): 1048.

Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R., & Gaston, K.J. (2011). Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. Journal of Applied Ecology, 48(5): 1125-1134.

Dawson, F.H. & Haslam, S.M. (1983). The management of river vegetation, with particular reference to shading effects of marginal vegetation. Landscape Planning, 10: 147-169.

Dawson, R. J., Ball, T., Werritty, J., Werritty, A., Hall, J. W. & Roche, N. (2011). Assessing the effectiveness of non-structural flood management measures in the Thames Estuary under conditions of socio-economic and environmental change. Global Environmental Change, 21: 628-646.

De Ruig, J. (1998). Coastline management in The Netherlands: human use versus natural dynamics. Journal of Coastal Conservation, 4(2): 127-134.

De Vita, P., Di Paolo, E., Fecondo, G., Di Fonzo, N., & Pisante, M. (2007). No-tillage and conventional tillage effects on durum wheat yield, grain quality and soil moisture content in southern Italy. Soil and Tillage Research, 92(1): 69-78.

De Vries, W., Leip, A., Reinds, G. J., Kros, J., Lesschen, J. P., & Bouwman, A. F. (2011). Comparison of land nitrogen budgets for European agriculture by various modeling approaches. Environmental Pollution, 159(11): 3254-3268.

DEFRA (2011). Biodiversity offsetting: Guiding principles for biodiversity offsetting, 9pp. Department for Environment, Food and Rural Affairs (DEFRA), London. Available: http://www.defra.gov.uk/environment/natural/biodiversity/uk/offsetting/ Accessed: August 2012.

Del Grosso, S.J., Ojima, D.S., Parton, W.J., Stehfest, E., Heistemann, M., DeAngelo, B., & Rose, S. (2009). Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global and Planetary Change, 67(1): 44-50.

DeLaune, R. & White, J. (2011). Will coastal wetlands continue to sequester carbon in response to an increase in global sea level?: a case study of the rapidly subsiding Mississippi river deltaic plain. Climatic Change 110(1-2): 297-314.

Delta Commissie (2008). Working together with Water. A living land builds for its future. Findings of the Deltacommissie. The Netherlands: Delta Commissie. Available: http://www.deltacommissie.com/doc/deltareport_full.pdf Accessed: June 2013.

Deng, X.P., Shan, L., Zhang, H.P., Turner, N.C. (2006) Improving agricultural water use efficiency in arid and semiarid areas of China. Agricultural Water Management, 80(1): 23-40.

Deng, Z.Y., Wang, Q., Zhang, Q., Qing, J.Z., Yang, Q.G., Yuan, Z.P., Liu, W.J., Xu, J.F. (2010). Impact of climate warming and drying on food crops in northern China and the countermeasures. Acta Ecologica Sinica 30(22):6278-6288. (in Chinese with English abstract)

Dennis, E.S., Dolferus, R., Ellis, M., Rahman, M., Wu, Y., Hoeren, F.U., Grover, A., Ismond, K.P., Good, A.G., Peacock, W.J. (2000). Molecular strategies for improving waterlogging tolerance in plants. J. Exp. Bot. 51 (342): 89-97.

DePauw, R.M., Knox, R.E., Clarke, F.R., Wang, H., Fernandez, M.R., Clarke, J.M., McCaig, T.N., 2007. Shifting undesirable correlations. Euphytica, 157: 409–415.

Dersch, G. and Böhm, K. (2001). Effects of agronomic practices on the soil carbon storage potential in arable farming in Austria. Nutrient Cycling in Agroecosystems 60(1): 49-55.

Desjardins, R.L., Smith, W., Grant, B., Campbell, C., & Riznek, R. (2005). Management strategies to sequester carbon in agricultural soils and to mitigate greenhouse gas emissions. Climatic Change, 70(1-2): 283-297.

Desrochers, D. W., Keagy, J.C., Cristol, D.A., (2008). Created versus natural wetlands: Avian communities in Virginia salt marshes. Ecoscience 15(1): 36-43.

Dessai, S. & Hulme, M. (2007). Assessing the robustness of adaptation decisions to climate change uncertainties: A case study on water resources management in the East of England. Global Environmental Change, 17(1): 59-72.

Ding, W., Cai, Z., & Wang, D. (2004). Preliminary budget of methane emissions from natural wetlands in China. Atmospheric Environment, 38(5): 751-759.

Dixon, A. M., Leggett, D.J., Weight, R.C., (1998). Habitat Creation Opportunities for Landward Coastal Re-alignment: Essex Case Studies. Water and Environment Journal, 12(2): 107-112.

Dixon, J. and Dupuis, A.N.N (2003). Urban Intensification in Auckland, New Zealand: A Challenge for New Urbanism. Housing Studies, 18(3): 353-368.

Dixon, M., Morris, R.K.A., Scott, C.R., Birchenough, A., Colclough, S., (2008). Managed Realignment - lessons from Wallasea, UK. Proceedings of the Institution of Civil Engineers - Maritime Engineering 161(2): 61-71.

Dixon, R., Brown, S., Houghton, R.E.A., Solomon, A.M., Trexler, M.C., & Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems. Science, 263(5144): 185-189.

Dodd, I.C., Whalley, W.R., Ober, E.S., & Parry, M.A.J. (2011). Genetic and management approaches to boost UK wheat yields by ameliorating water deficits. Journal of Experimental Botany, 62(15): 5241-5248.

Dodson, J. (2010). In the Wrong Place at the Wrong Time? Assessing some Planning, Transport and Housing Market Limits to Urban Consolidation Policies. Urban Policy and Research 28(4): 487-504.

Doick, K., Hutchings, T. (2013). Air temperature regulation by urban trees and green infrastructure, The Forestry Commission, Forest Research, FCRN012, 10pp.

Domènech, L., & Saurí, D. (2011). A comparative appraisal of the use of rainwater harvesting in single and multi-family buildings of the Metropolitan Area of Barcelona (Spain): social experience, drinking water savings and economic costs. Journal of Cleaner Production, 19(6), 598-608.

Du, K.Y., Shi, S.J., Zheng, D.W. (2009). Comprehensive impacts of adaptation measures to climate change based on maize production in Jilin Province. Chinese Journal of Agrometeorology 30(1):6-9 (in Chinese with English abstract)

Du, T.S., Kang, S.Z., Sun, J.S., Zhang, X., Zhang, J. (2010) An improved water use efficiency of cereals under temporal and spatial deficit irrigation in north China. Agricultural Water Management, 97: 66-74.

Durandeau, S., Gabrielle, B., Godard, C., Jayet, P.A., & Le Bas, C. (2010). Coupling biophysical and micro-economic models to assess the effect of mitigation measures on greenhouse gas emissions from agriculture. Climatic change, 98(1-2): 51-73.

Dyke, E. V. and K. Wasson (2005). Historical Ecology of a Central California Estuary: 150 Years of Habitat Change. Estuaries 28(2): 173-189.

Easterling, W.E., Aggarwal, P.K., Batima, P., Brander, K.M., Erda, L., Howden, S. M., Kirilenko, A., Morton, J., Soussana, J.-F., Schmidhuber, J. & Tubiello, F.N. (2007). Food, fibre and forest products. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & HANSON, C.E. (Eds.) Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

Easterling, W.E., Mearns, L.O., Hays, C.J., & Marx, D. (2001). Comparison of agricultural impacts of climate change calculated from high and low resolution climate change scenarios: Part II. Accounting for adaptation and CO2 direct effects. Climatic Change, 51(2): 173-197.

ECCP (2001). Mitigation potential of greenhouse gases in the agricultural sector. European Climate Change Programme, Working Group 7 – agriculture final report, 50pp.

Eertman, R.H.M., Kornman, B.A., Stikvoort, E., Verbeek, H., (2002). Restoration of the Sieperda tidal marsh in the Scheldt Estuary, the Netherlands. Restoration Ecology, 10(3): 438-449.

Elgershuizen, J.H.B.W. (1981). Some environmental impacts of a storm surge barrier. Marine Pollution Bulletin, 12(8): 265-271.

Emmerson, R.H.C., Manatunge, J.M.A., MacLeod, C.L., Lester, J.N., (1997). Tidal Exchanges Between Orplands Managed Retreat Site and the Blackwater Estuary, Essex. Water and Environment Journal, 11(5): 363-372.

Emmerson, R.H.C., Birkett, J.W., Scrimshaw, M., Lester, J.N., (2000). Solid phase partitioning of metals in managed retreat soils: field changes over the first year of tidal inundation. The Science of the Total Environment, 254(1): 75-92.

Enemark, J. (2005). The Wadden Sea protection and management scheme - towards an integrated coastal management approach? Ocean & Coastal Management 48(11-12): 996-1015.

Engle, N. L., de Bremond, A., Malone, E. L., & Moss, R. H. (2013). Towards a resilience indicator framework for making climate-change adaptation decisions. Mitigation and Adaptation Strategies for Global Change, 1-18.

Escobedo, F., Varela, S., Zhao, M., Wagner, J. E., & Zipperer, W. (2010). Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. Environmental Science & Policy, 13(5): 362-372.

Etterson, J.R., 2008. Evolution in response to climate change. In: Carroll, S.P, & Fox, C.W. (Eds.), Conservation Biology: Evolution in Action. Oxford University Press, New York.

European Commission (2011). Cities of tomorrow: Challenges, visions, way forward. European Commission, Directorate General for Regional Policy. Urban Development, Territorial Cohesion unit, 116pp.

European Environment Agency (2007). Climate change and water adaptation issues. Luxembourg: Office for Official Publications of the European Communities. Avaliable: http://www.google.co.uk/url?sa=t&rct=j&q=climate%20change%20and%20water%20adaptation%20issues&source=web&cd=3&ved=0CFsQFjAC&url=http%3A%2F%2Fwww.eea.europa.eu%2Fpublications%2Ftechnical_report_2007_2%2Fat_download%2Ffile&ei=N4T1T8-mHKii0QW715C1Bw&usg=AFQjCNGOSzoOX79cjJ5ZwWopuIKcOp7MGg Accessed: June 2012.

European Environment Agency (2012). Environmental Indicator Report 2012. Ecosystem resilience and resource efficiency in a green economy in Europe. Copehagen: European Environment Agency. Available: http://www.eea.europa.eu/publications/environmental-indicator-report-2012/environmental-indicator-report-2012-ecosystem/authors.xhtml Accessed: July 2013.

Evans, E., Ashely, R., Hall, J., Penning-Rowsell, E., Sayers, P., Thorne, C. & Watkinson, A. 2004. Foresight future flooding. Volume 1. Future risks and their drivers. London: Office of Science and Technology.

Falk, K., Nøhr, H., Rasmussen, L.M., (1994). Margrethe-Kog and the Artificial Saltwater Lagoon: Evaluation of a Habitat Restoration Project in the Danish Wadden Sea. Environmental Conservation, 21(2): 133-144.

Falloon, P. & Betts, R. (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.

Falloon, P., Smith, P., Betts, R., Jones, C. D., Smith, J., Hemming, D., & Challinor, A. (2009). Carbon Sequestration and Greenhouse Gas Fluxes from Cropland Soils–Climate Opportunities and Threats. In Climate Change and Crops (pp. 81-111). Springer Berlin Heidelberg.

Fankhauser, S., Burton, I. (2011). Spending adaptation money wisely. Climate Policy 11(3): 1037-1049.

FAO (2010). Managing forests for climate change, I1960E, 20pp.

FAO (2012). Forests and climate change working paper 10: Forest management and climate change: a literature review. MD012/E, 55pp

Farley, K. A., Jobbágy, E. G., Jackson, R. B. (2005), Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology, 11: 1565–1576.

Farreny, R., Gabarrell, X., & Rieradevall, J. (2011). Cost-efficiency of rainwater harvesting strategies in dense Mediterranean neighbourhoods. Resources, conservation and recycling, 55(7): 686-694.

Fernandez-Canero, R., & Gonzalez-Redondo, P. (2010). Green roofs as a habitat for birds: a review. Journal of Animal and Veterinary Advances, 9(15): 2041-2052.

Few, R., Brown, K., Tompkins, E.L. (2004). Scaling adaptation: climate change response andcoastal management in the UK. Working Paper 60, Tyndall Centre for Climate Change Research, University of East Anglia, Norwich.

Field, R.H., Benke, S., Bádonyi, K., & Bradbury, R.B. (2007). Influence of conservation tillage on winter bird use of arable fields in Hungary. Agriculture, ecosystems & environment, 120(2): 399-404.

Filipovic, D., Kosutic, S., Gospodaric, Z., Zimmer, R., & Banaj, D. (2006). The possibilities of fuel savings and the reduction of CO2 emissions in the soil tillage in Croatia. Agriculture, ecosystems & environment, 115(1): 290-294.

Finger, R., Hediger, W., & Schmid, S. (2011). Irrigation as adaptation strategy to climate change - a biophysical and economic appraisal for Swiss maize production. Climatic change, 105(3-4): 509-528.

Fioretti, R., Palla, A., Lanza, L.G., & Principi, P. (2010). Green roof energy and water related performance in the Mediterranean climate. Building and Environment, 45(8): 1890-1904.

Fischer, G., Tubiello, F.N., van Velthuizen, H. & Wiberg, D.A. (2007). Climate change impacts on irrigation water requirements: Effects of mitigation, 1990–2080. Technological Forecasting and Social Change, 74: 1083-1107.

Florides, G.A., Kalogirou, S.A., Tassou, S.A., & Wrobel, L.C. (2000). Modeling of the modern houses of Cyprus and energy consumption analysis. Energy, 25(10): 915-937.

Fornara, D.A., Steinbeiss, S., McNamara, N.P., Gleixner, G., Oakley, S., P. R. Poulton, P.R., Macdonald, A.J., Bardgett, R.D. (2011). Increases in soil organic carbon sequestration can reduce the global warming potential of long-term liming to permanent grassland. Global Change Biology, 17(5): 1925-1934.

Foulkes, M.J., Snape, J.W., Shearman, V.J., Reynolds, M.P., Gaju, O., & Sylvester-Bradley, R. (2007). Genetic progress in yield potential in wheat: recent advances and future prospects. Journal of Agricultural Science, 145(1): 17.

Franzluebbers, A. J. (2005). Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil and Tillage Research, 83(1): 120-147.

Fraser, A.F., & Broom, D.M. (1997). *Farm animal behaviour and welfare* (No. Ed. 3). CAB international, 437pp.

Freibauer, A., Rounsevell, M. D. A., Smith, P. & Verhagen, J. 2004. Carbon sequestration in the agricultural soils of Europe. Geoderma, 122, 1-23.

French, J.R. (2008). Hydrodynamic modelling of estuarine flood defence realignment as an adaptive management response to sea-level rise. Journal of Coastal Research 24(2A): 1-12.

French, P.W. (1999). Managed retreat: a natural analogue from the Medway estuary, UK. Ocean & Coastal Management 42(1): 49-62.

Fryd, O., Jensen, M.B., Ingvertsen, S.T., Jeppesen, J., & Magid, J. (2010). Doing the first loop of planning for sustainable urban drainage system retrofits: A case study from Odense, Denmark. Urban Water Journal, 7(6): 367-378.

Gaiser, T., Abdel-Razek, M. & Bakara, H. (2009). Modeling carbon sequestration under zero-tillage at the regional scale. II. The influence of crop rotation and soil type. Ecological Modelling, 220: 3372-3379.

Gao, W.H., Deng, X.P. (2007). Techniques of effective conservation and efficient use of water on terraced land in Yan'an city. Bulletin of Soil and Water Conservation, 27: 225-229.

Garbutt, A. and Wolters, M. (2008). The natural regeneration of salt marsh on formerly reclaimed land. Applied Vegetation Science, 11(3): 335-344.

Garbutt, R.A., Reading, C.J., Wolters, M., Gray, A.J., Rothery, P., (2006). Monitoring the development of intertidal habitats on former agricultural land after the managed realignment of coastal defences at Tollesbury, Essex, UK. Marine Pollution Bulletin 53(1-4): 155-164.

Gardiner, S.H., S., Nicholls, R. , Zhang, Z., Jude, S. , Jones, A., Richards, J., Williams, A., Spencer, T., Cope, S., Gorczynska, M., Bradbury, A., McInnes, R., Ingleby, A., Dalton, H. (2007). The habitats directive, coastal habitats and climate change - Case studies from the south coast of the UK. Southampton Tyndall Centre for Climate Change Research, 17pp.

Garforth M. (2012). Adaptation of forests to climate change - Report of desk-based research on resilience of forests to climate change and transformation measures. Dzneladze, M. (ed.). Report prepared for the EU funded Project DCI-ENV/2010/221391. WWF Caucasus Programme Office, Tbilisi.

Ge, D.K., Jin, Z.Q., Shi, C.L., Gao, L.Z. (2002). Gradual impacts of climate change on rice production and adaptation strategies in Southern China. Jiangsu J of Agr Sci, 18(1):1-8.

Genskow, K. D. (2012). Taking stock of voluntary nutrient management: Measuring and tracking change. Journal of Soil and Water Conservation, 67(1): 51-58.

Gerakis, A., & Kalburtji, K. (1998). Agricultural activities affecting the functions and values of Ramsar wetland sites of Greece. Agriculture, ecosystems & environment, 70(2): 119-128.

Getter, K.L., Rowe, D.B., Robertson, G.P., Cregg, B.M., & Andresen, J.A. (2009). Carbon sequestration potential of extensive green roofs. Environmental science & technology, 43(19): 7564-7570.

Giacomini, S.J., Machet, J.M., Boizard, H. & Recous, S. (2010). Dynamics and recovery of fertilizer 15N in soil and winter wheat crop under minimum versus conventional tillage. Soil and Tillage Research, 108(1): 51-58.

Giannakopoulos, C., Le Sager, P., Bindi, M., Moriondo, M., Kostopoulou, E., & Goodess, C.M. (2009). Climatic changes and associated impacts in the Mediterranean resulting from a 2 C global warming. Global and Planetary Change, 68(3): 209-224.

Gifford, R.M., Campbell, B.D., & Howden, S.M. (1996). Options for adapting agriculture to climate change: Australian and New Zealand examples. Eds Bouma, W.J., Pearman, G.I., Manning, M.R. Greenhouse: coping with climate change. CSIRO Australia, 399-416.

Gill, S.E., Handley, J.F., Ennos, A.R., Pauleit, S., (2007). Adapting cities for climate change: the role of the green infrastructure. Built Environment, 33(1): 115-133.

Giller, K. E. and Cadisch, G. (1995). Future benefits from biological nitrogen fixation: An ecological approach to agriculture. Plant and Soil, 174(1): 255-277.

GLA (2010). The draft climate change adaptation strategy for London. Public consultation draft. Greater London Authority, 138pp. Available: http://legacy.london.gov.uk/mayor/priorities/docs/Climate_change_adaptation_080210.pdf Accessed: July 2013.

Glasby, T., Connell, S.D., Holloway, M.G., Hewitt, C.L., (2007). Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? Marine Biology, 151(3): 887-895.

Glendenning, C.J. & Vervoort, R.W. (2011). Hydrological impacts of rainwater harvesting (RWH) in a case study catchment: The Arvari River, Rajasthan, India: Part 2. Catchment-scale impacts. Agricultural Water Management, 98: 715-730.

Gómez-Limón, J.A. and Riesgo, L. (2004). Irrigation water pricing: differential impacts on irrigated farms. Agr. Econ., 31: 47–66.

Gomez-Ullate, E., Castillo-Lopez, E., Castro-Fresno, D., & Bayon, J.R. (2011). Analysis and contrast of different pervious pavements for management of storm-water in a parking area in northern Spain. Water resources management, 25(6): 1525-1535.

Gorantiwar, S.D. and Smout, K. (2003). Allocation of scarce water resources using deficit irrigation in rotational systems. Journal of Irrigation and Drainage Engineering, 3: 155–163.

Gordon, R.J., Vanderzaag, A.C., Dekker, P.A., De Haan, R. & Madani, A. (2011). Impact of modified tillage on runoff and nutrient loads from potato fields in Prince Edward Island. Agricultural Water Management, 98: 1782-1788.

Górny, A. G., Banaszak, Z., Ługowska, B., & Ratajczak, D. (2011). Inheritance of the efficiency of nitrogen uptake and utilization in winter wheat (Triticum aestivum L.) under diverse nutrition levels. Euphytica, 177(2): 191-206.

Götting, E. (2001). Development of salt marsh arthropod fauna after opening a summer polder. Marine Biodiversity, 31(2): 333-340.

Graddon, A.R., Kuczera, G., & Hardy, M.J. (2011). A flexible modelling environment for integrated urban water harvesting and re-use. Water Science & Technology, 63(10): 2268-2278.

Grayling, T. (2009). More climate change adaptation for less emissions. IEMA environmental assessment forum, London, UK.

Gregorich, E.G., Rochette, P., VandenBygaart, A.J., & Angers, D.A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil and Tillage Research, 83(1): 53-72.

Grippo, M.A., Cooper, S., Massey, A.G., (2007). Effect of Beach Replenishment Projects on Waterbird and Shorebird Communities. Journal of Coastal Research, 23(5): 1088-1096.

Groover, A.T. (2007). Will genomics guide a greener forest biotech? Trends in Plant Science, 12 (6): 234-238.

Guan, P. (2004). Analysis of the efficiency of water-saving irrigation and project benefit of the secondly irrigation district in Jingdian. Water Saving Irrigation, 5:68-71.

Guidetti, P., Bussotti, S., Boero, F., (2005). Evaluating the effects of protection on fish predators and sea urchins in shallow artificial rocky habitats: a case study in the northern Adriatic Sea. Marine Environmental Research, 59(4): 333-348.

Guo, Q.Y., Huang, G.B., Li, G.D. (2005). Conservation tillage effects on soil moisture and water use efficiency of two phases rotation system with spring wheat and field pea in dryland. Journal of Soil and Water Conservation, 19(3):165-169.

Gupta, A.K., Tesluk, P.E., & Taylor, M.S. (2007). Innovation at and across multiple levels of analysis. Organization Science, 18(6): 885-897.

Gupta, G.S., Shukla, S. P., Prasad, G., & Singh, V.N. (1992). China clay as an adsorbent for dye house wastewaters. Environmental technology, 13(10): 925-936.

Habash, D.Z., Kehel, Z. & Nachit, M. (2009). Genomic approaches for designing durum wheat ready for climate change with a focus on drought. Journal of Experimental Botany, 60: 2805–2815.

Hall, J.M., Handley, J.F., & Ennos, A.R. (2012). The potential of tree planting to climate-proof high density residential areas in Manchester, UK. Landscape and Urban Planning, 104(3): 410-417.

Hallegatte, S. (2009). Strategies to adapt to an uncertain climate change. Global Environmental Change, 19: 240-247.

Hampel, H., Cattrijsse, A., Vincx, M., (2003). Habitat value of a developing estuarine brackish marsh for fish and macrocrustaceans. ICES Journal of Marine Science: Journal du Conseil, 60(2): 278-289.

Hannah, L. (2010). A Global Conservation System for Climate-Change Adaptation. Conservation Biology, 24(1): 70-77.

Hannah, L., Midgley, G., Andelman, S., Araújo, M., Hughes, G., Martinez-Meyer, E., Pearson, R., Williams, P. (2007. Protected area needs in a changing climate. Frontiers in Ecology and the Environment 5(3): 131-138.

Hannah, L., Midgley, G., Lovejoy, T., Bond, W.J., Bush, M.L.J.C., Lovett, J.C., Scott, D., Woodward, F..I. (2002). Conservation of biodiversity in a changing climate. Conservation Biology, 16(1): 264-268.

Hansen, L.T. (2009). The viability of creating wetlands for the sale of carbon offsets. Journal of Agricultural and Resource Economics, 34(2): 350-365.

Hansen. J.W., Challinor, A., Ines, A., Wheeler, T., Moron, V. (2006). Translating climate forecasts into agricultural terms: advances and challenges. Clim. Res. 33: 27-41.

Hansford, M. (2004). Flooding fears push agency to seek jubilee river extension. New Civil Engineer, 13th May 2004.

Hardaway Jr, C.S. and J.R. Gunn (2010). Design and performance of headland bays in Chesapeake Bay, USA. Coastal Engineering, 57(2): 203-212.

Harmin, E.M., & Gurran, N. (2009). Urban form and climate change: balancing adaptation and mitigation in the U.S. and Australia. Habitat International, 33: 238-245

Harrison, P.A., Holman, I.P., Cojocaru, G., Kok, K., Kontogianni, A., Metzger, M.J., Gramberger, M., 2012, Combining qualitative and quantitative understanding for exploring cross-sectoral climate change impacts, adaptation and vulnerability in Europe. Regional Environmental Change, 1-20.

Harrison, P.A., Porter, J.R., Downing, T.E. (2000). Scaling-up the AFRC-WHEAT2 model to assess phenological development for wheat in Europe. Agric. Forest Meteorol. 101: 167–186.

Harry, S., and Morad, M. (2013). Sustainable development and climate change: Beyond mitigation and adaptation. Local Economy 28(4): 358-368.

Hartley, M.J. (2002). Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management, 155(1): 81-95.

Hayek, M., Arku, G., Gilliland, J. (2010). Assessing London, Ontario's brownfield redevelopment effort to promote urban intensification. Local Environment 15(4): 389-402.

Hazelden, J., Boorman, L.A. (2001). Soils and ‘managed retreat’ in South East England. Soil Use and Management, 17(3): 150-154.

Hegedüs, A., Gaál, M., Bérces, R. (2011). Tree appraisal methods and their application – first results in one of Budapest’s districts. Applied Ecology and Environmental Research, 9(4): 411-423.

Hellegers, P., Zilberman, D., & van Ierland, E. (2001). Dynamics of agricultural groundwater extraction. Ecological Economics, 37(2): 303-311.

Heller, N.E., & Zavaleta, E.S. (2009). Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biological conservation, 142(1): 14-32.

Henry, B., Eckard, R., Gaughan, J.B., & Hegarty, R. (2012). Livestock production in a changing climate: adaptation and mitigation research in Australia. Crop and Pasture Science, 63(3): 191-202.

Herrero, M., Thornton, P.K., Gerber, P., Reid, R.S. (2009). Livestock, livelihoods and the environment: understanding trade-offs. Current opinion in environmental sustainability, 1(2): 111-120.

Herridge, D.F., Peoples, M.B., & Boddey, R.M. (2008). Global inputs of biological nitrogen fixation in agricultural systems. Plant and Soil, 311(1-2): 1-18.

Hillier, J., Brentrup, F., Wattenbach, M., Walter, C., Garcia-Suarez, T., Mila-i-Canals, L., & Smith, P. (2012). Which cropland greenhouse gas mitigation options give the greatest benefits in different world regions? Climate and soil-specific predictions from integrated empirical models. Global Change Biology, 18(6): 1880-1894.

Hinkle, R.L., Mitsch, W.J. (2005). Salt marsh vegetation recovery at salt hay farm wetland restoration sites on Delaware Bay. Ecological Engineering, 25(3): 240-251.

Hobbs, P. R. (2007). Conservation agriculture: what is it and why is it important for future sustainable food production? Journal of Agricultural Science, 145(2): 127.

Hodgson, J.A., Thomas, C.D., Cinderby, S., Cambridge, H., Evans, P., & Hill, J.K. (2011). Habitat re-creation strategies for promoting adaptation of species to climate change. Conservation Letters, 4(4): 289-297.

Hoegh-Guldberg, O., Hughes, L., McIntyre, S., Lindenmayer, D.B., Parmesan, C., Possingham, H.P., & Thomas, C.D. (2008). Assisted colonization and rapid climate change. Science (Washington), 321(5887): 345-346.

Hoes, P., Trcka, M., Hensen, J.L.M., & Bonnema, B.H. (2011). Optimizing building designs using a robustness indicator with respect to user behavior. In Proceedings of the 12th Conference of the International Building Performance Simulation Association, Sydney (pp. 1710-1717).

Hofstede, J.L.A. (2003). Integrated management of artificially created salt marshes in the Wadden Sea of Schleswig-Holstein, Germany. Wetlands Ecology and Management, 11(3): 183-194.

Holmes, M.J., & Hacker, J.N. (2007). Climate change, thermal comfort and energy: Meeting the design challenges of the 21st century. Energy and Buildings, 39(7): 802-814.

Holmgren, P., Marklund, L-G., Saket, M. & Wilkie, M.L. (2007). Forest Monitoring and Assessment for Climate Change Reporting: Partnerships, Capacity Building and Delivery. Forest Resources Assessment Working Paper 142. FAO, Rome, 48pp.

Hoque, M. & Alam, M. (1997). Subsidence in the lower deltaic areas of Bangladesh. Marine Geodesy, 20: 105-120.

Hossler, K., & Bouchard, V. (2010). Soil development and establishment of carbon-based properties in created freshwater marshes. Ecological Applications, 20(2): 539-553.

Howden, S.M., Soussana, J.-F., Tubiello, F.N., Chhetri, N., Dunlop, M., Meinke, H. (2007). Adapting agriculture to climate change, Proceedings of the National Academy of Sciences, 104(50): 19691-19696.

Howgate, O.R. & Kenyon, W. (2009). Community cooperation with natural flood management: a case study in the Scottish Borders. Area, 41(3): 329-340.

Howley, P. (2009). Attitudes towards compact city living: Towards a greater understanding of residential behaviour. Land Use Policy, 26(3): 792-798.

Høyer, K.G. (2009). Combining stationary and mobile energy requirements: the importance of the meso-level in sustainable urban development. Indoor and Built Environment, 18(5): 407-415.

Hughes, G., Chinowsky, P. & Strzepek, K. (2010). The costs of adaptation to climate change for water infrastructure in OECD countries. Utilities Policy, 18: 142-153.

Hulme, M., Adger, W.N., Dessai, E., Goulden, M., Lorenzoni, I., Nelson, D., Naess, L.-O., Wolf, J., Wreford A. (2007). Limits and barriers to adaptation: four propositions. Tyndall Centre for Climate Change Research, Tyndall Briefing Note No. 20

Hunde, A., Patterson, P., Ricke, S., & Kim, W.K. (2012). Supplementation of Poultry Feeds with Dietary Zinc and Other Minerals and Compounds to Mitigate Nitrogen Emissions—A Review. Biological Trace Element Research, 147(1-3): 386-394.

Hunter, M.L. (2007). Climate change and moving species: furthering the debate on assisted colonization. Conservation Biology, 21(5): 1356-1358.

Huntjens, P., Lebel, L., Pahl-Wostl, C., Camkin, J., Schulze, R., & Kranz, N. (2012). Institutional design propositions for the governance of adaptation to climate change in the water sector. Global Environmental Change, 22(1): 67-81.

Huq S, Grubb, M. (2007) Preface: Mitigation and Adaptation Strategies for Global Change Springer, 12(5): 645–649

Iglesias, A., Garrote, L., Diz, A., Schlickenrieder, J. & Martin-Carrasco, F. (2011). Re-thinking water policy priorities in the Mediterranean region in view of climate change. Environmental Science and Policy, 14: 744-757.

Iglesias, A., Garrote, L., Flores, F., Moneo, M. (2007). Challenges to manage the risk of water scarcity and climate change in the Mediterranean. Water Resource Management, 21: 775-788.

Ip, K., & Miller, A. (2009). Thermal behaviour of an earth-sheltered autonomous building–The Brighton Earthship. Renewable Energy, 34(9): 2037-2043.

Irving, A. D., Connell, Sean D. and Russell, Bayden D. (2011). Restoring Coastal Plants to Improve Global Carbon Storage: Reaping What We Sow. PLoS One 6(3): e18311.

Ivey, J.L., Smithers, J., de Loë, R.C., & Kreutzwiser, R.D. (2004). Community capacity for adaptation to climate-induced water shortages: linking institutional complexity and local actors. Environmental Management, 33(1): 36-47.

Jackson, N.L., Jackson, N.L., Nordstrom, K.F., Saini, S., Smith, D.R., (2010). Effects of nourishment on the form and function of an estuarine beach. Ecological Engineering 36(12): 1709-1718.

Jackson, R.B., Jobbagy, E.G., Avissar, R., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., le Maitre, D.C., McCarl, B.A., Murray, B.C. (2005). Trading water for carbon with biological carbon sequestration. Science, 310(5756): 1944-1947.

Jackson, R.B., Randerson, J.T., Canadell, J.G., Anderson, R.G., Avissar, R., Baldocchi, D.D., Bonan, G.B., Caldeira, K., Diffenbaugh, N.S., Field, C.B., Hungate, B.A., Jobbagy, E.G., Kueppers, L.M., Nosetto, M.D., and Pataki, D.E. (2008). Protecting climate with forests. Environ. Res. Lett. 3(4): 044006.

Jaffal, I., Ouldboukhitine, S. E., & Belarbi, R. (2012). A comprehensive study of the impact of green roofs on building energy performance. Renewable Energy, 43: 157-164.

James, E. and Baldani, J. (2012). The role of biological nitrogen fixation by non-legumes in the sustainable production of food and biofuels. Plant and Soil, 356(1): 1-3.

Janousek, C.N., Currin, C.A., Levin, L.A., (2007). Succession of microphytobenthos in a restored coastal wetland. Estuaries and Coasts, 30(2): 265-276.

Jarvis, A., Lau, C., Cook, S., Wollenberg, E., Hansen, J., Bonilla, O., Challinor, A. (2011). An integrated adaptation and mitigation framework for developing agricultural research: Synergies and trade-offs. Expl. Agric. 47(2): 185-203.

Jefferson, I., Rogers, C.D.F., & Hunt, D.V.L. (2006). Achieving sustainable underground construction in Birmingham Eastside. In 10th Congress of the International Association for Engineering Geology and the Environment (IAEG), Nottingham, UK.

Jensen, E.S., Peoples, M.B., Boddey, R.M., Gresshoff, P.M., Hauggaard-Nielsen, H., Alves, B.J., & Morrison, M.J. (2012). Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries. A review. Agronomy for sustainable development, 32(2): 329-364.

Jickells, T., Andrews, J., Cave, R., and Parkes, D. (2003). biogeochemical value of intertidal areas – a case study of the Humber Estuary. Wetland valuation: state of the art and opportunities for further development, 125pp.

Jin, D.C., Gao, G.L. (2012). A Brief Discussion on Basic Aspects of Agricultural Pest Control under Global Climate Change. Journal of Mountain Agriculture and Biology, 31(1): 63-69.

Jin, L., Li, Y.E., Gao, Q.Z., Liu, Y.T., Wan, Y.F., Qin, X.B., Shi, F. (2008). Estimate of carbon sequestration under cropland management in China. Scientia Agricultura Sinica, 41(3): 734-743.

Jin, Z.Q., Ge, D.K., Gao, L.Z., Shi, C.L. (1998). Food crop production strategies for adaptation to global climate change in the sampling zones of Eastern China-A modelling study. Scientia Agricultura Sinica, 31(4): 51-58.

Johnston, M., Williamson, T., Munson, A., Ogden, A., Moroni, M., Parsons, R. Price D., Stadt, J. (2010). Climate change and forest management in Canada: impacts, adaptive capacity and adaptation options. A state of knowledge report. Sustainable Forest Management network, Edmonton, Alberta, 54pp.

Joint Research Centre (2012). Science for Water. JRC thematic report. Brussels: European Commission. Available: http://ec.europa.eu/dgs/jrc/downloads/jrc_2012_science_for_water_en.pdf Accessed: June 2012.

Jong, D.J., and Kogel, T.J. (1985). Salt-marsh research with respect to the civil-engineering project in the Oosterschelde, the Netherlands. Plant Ecology 62(1): 425-432.

Jordan, E.R., (2003). Effects of heat stress on reproduction. Journal of Dairy Science, 86(13 Supplement): E104-E114.

Kairo, J.G., Dahdouh-Guebas, F., Bosire, J. and Koedam, N. (2001). Restoration and management of mangrove systems- a lesson for and from the East African region. South African Journal of Botany, 67: 383–389.

Kaparaju, P. and Rintala, J. (2011). Mitigation of greenhouse gas emissions by adopting anaerobic digestion technology on dairy, sow and pig farms in Finland. Renewable Energy, 36(1): 31-41.

Kassam, A., Friedrich, T., Derpsch, R., Lahmar, R., Mrabet, R., Basch, G., González-Sánchez, E.J. & Serraj, R. (2012). Conservation agriculture in the dry Mediterranean climate. Field Crops Research, 132: 7-17.

Kazmierczak, A. (2012). Heat and social vulnerability in Greater Manchester: a risk-response case study. EcoCities Project, University of Manchester, Manchester, UK.

Kazmierczak, A., & Carter, J. (2010). Adaptation to climate change using green and blue infrastructure. A database of case studies. Interreg IVC Green and blue space adaptation for urban areas and eco towns (GRaBS). Manchester, UK. Available at: http://www.grabs-eu.org/ Accessed: July 2012.

Kellomäki, S., & Leinonen, S. (Eds), (2005). SilviStrat Final Report, Silvicultural Response Strategies to Climatic Change in Management of European Forests, Funded by the European Union Under the Contract EVK2-2000-00723 (SilviStrat). Available online: http://www.efi.int/portal/research/research_programmes/sustainability_and_climate_change/silvistrat/ Accessed: July 2013.

Kellomäki, S., and Leinonen, S. (2005). Final Report of the Project "Silvicultural Response Strategies to Climatic Change in Management of European Forests" funded by the European Union under the Contract EVK2-2000- 00723 (SilviStrat).

Kennedy, I.R., Pereg-Gerk, L.L., Wood, C., Deaker, R., Gilchrist, K., & Katupitiya, S. (1997). Biological nitrogen fixation in non-leguminous field crops: facilitating the evolution of an effective association between Azospirillum and wheat. Plant and soil, 194(1-2): 65-79.

Keppel, G., van Niel, K.P., Wardell-Johnson, G.W., Yates, C.J., Byrne, M., Mucina, L., Schut, A.G.T., Hopper, S.D., Franklin, S.E. (2012). Refugia: identifying and understanding safe havens for biodiversity under climate change. Global Ecology and Biogeography 21(4): 393-404.

Kerr, N.J., Siddique, K.H.M., & Delane, R.J. (1992). Early sowing with wheat cultivars of suitable maturity increases grain yield of spring wheat in a short season environment. Animal Production Science, 32(6): 717-723.

Khaledian, M.R., Mailhol, J.C., Ruelle, P., Mubarak, I., & Perret, S. (2010). The impacts of direct seeding into mulch on the energy balance of crop production system in the SE of France. Soil and Tillage Research, 106(2): 218-226.

King, J.A., Bradley, R.I., Harrison, R., & Carter, A.D. (2004). Carbon sequestration and saving potential associated with changes to the management of agricultural soils in England. Soil Use and Management, 20(4): 394-402.

King, S. E. and J. N. Lester (1995). The value of salt marsh as a sea defence. Marine Pollution Bulletin, 30(3): 180-189.

Kirda, C., Moutonnet, P., Hera, C., Nielsen, D.R. (1999). Crop yield response to deficit irrigation. In: Developments in Plant and Soil Sciences, 84, Kluwer Academic Publishers, Dordrecht, Netherlands.

Kirshen, P. (2007). Adaptation options and cost in water supply. Detailed analysis by sectors prepared for background paper, report to the UNFCCC. Available: http://unfccc.int/files/cooperation_and_support/financial_mechanism/application/pdf/kirshen.pdf Accessed: June 2012

Kirwan, M.L., and Blum, L.K. (2011). Enhanced decomposition offsets enhanced productivity and soil carbon accumulation in coastal wetlands responding to climate change. Biogeosciences, 8(4): 987-993.

Klausmeyer, K.R., & Shaw, M.R. (2009). Climate change, habitat loss, protected areas and the climate adaptation potential of species in Mediterranean ecosystems worldwide. PLoS One, 4(7): e6392.

Kleerekoper, L., van Esch, M., Salcedo, T.B. (2012). How to make a city climate-proof: Addressing the urban heat island effect. Resources, Conservation and Recycling, 64: 30-38.

Klein, R. J. T., Smit, M.J., Goosen, H., Hulsbergen, C.H. (1998). Resilience and Vulnerability: Coastal Dynamics or Dutch Dikes? The Geographical Journal, 164(3): 259-268.

Klein, R.J.T. and Bateman, I.J. (1998). The recreational value of Cley marshes nature reserve: an argument against managed retreat? Water and Environment Journal, 12(4): 280-285.

Klein, R.J.T., Huq, S., Denton, F., Downing, T.E., Richels, R.G., Robinson, J.B., Toth, F.L. (2007). Inter-relationships between adaptation and mitigation. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J., and Hanson, C.E. (Eds.) Cambridge University Press, Cambridge, UK, 745-777.

Klein, R.J.T., Nicholls, R.J., Ragoonaden, S., Capobianco, M., Aston, J., Buckley, E.N., (2001). Technological Options for Adaptation to Climate Change in Coastal Zones. Journal of Coastal Research, 17(3): 531-543.

Klijn, F., van Buuren, M. & van Rooij, S.A.M. (2004). Flood-risk management strategies for an uncertain future: living with Rhine river floods in the Netherlands? Ambio, 33: 141-147.

Klik, A., & Eitzinger, J. (2010). Impact of climate change on soil erosion and the efficiency of soil conservation practices in Austria. The Journal of Agricultural Science, 148(5): 529-541.

Kolditz, K., Dellwig, O., Barkowski, J., Beck, M., Freund, H., Brumsack, H.-J., (2009). Effects of de-embankment on pore water geochemistry of salt marsh sediments. Journal of Coastal Research, 25(6): 1222-1235.

Kolström, M., Lindner, M., Vilén, T., Maroschek, M., Seidl, R., Lexer, M.J., Netherer, S., Kremer, A., Delzon, S., Barbati, A., Marchetti, M., and Corona, P. (2011). Reviewing the science and implementation of climate change: adaptation measures in European forestry. Forests, 2: 961-982.

Krausse, B., Cook, M.J., Lomas, K.J. (2007). Environmental performance of a naturally ventilated city centre library. Energy and Buildings, 39(7): 792-801.

Kreyling, J., Bittner, T., Jaeschke, A., Jentsch, A., Jonas Steinbauer, M., Thiel, D., & Beierkuhnlein, C. (2011). Assisted colonization: a question of focal units and recipient localities. Restoration Ecology, 19(4): 433-440.

Kros, J., de Vries, W., Reinds, G.J., Lesschen, J.P., & Velthof, G.L. (2010). Assessment of the impact of various mitigation options on nitrous oxide emissions caused by the agricultural sector in Europe. Journal of Integrative Environmental Sciences, 7(S1): 223-234.

Krosby, M., Tewksbury, J., Haddad, N.M., & Hoekstra, J. (2010). Ecological connectivity for a changing climate. Conservation Biology, 24(6): 1686-1689.

Kundzewicz, Z.W., Mata, L.J., Arnell, N.W., Döll, P., Kabat, P., Jiménez, B., Miller, K.A., Oki, T., Sen, Z., Shiklomanov, I.A. (2007). Freshwater resources and their management. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J., Hanson, C.E., (Eds.) Cambridge University Press, Cambridge, UK, 173-210.

Lafortezza, R., Carrus, G., Sanesi, G., & Davies, C. (2009). Benefits and well-being perceived by people visiting green spaces in periods of heat stress. Urban Forestry & Urban Greening, 8(2): 97-108.

Lal, R. (2004). Soil carbon sequestration to mitigate climate change. Geoderma, 123(1): 1-22.

Lal, R. (2005). Forest soils and carbon sequestration. Forest Ecology and Management, 220(1): 242–258.

Lamb, D. (1998). Large scale ecological restoration of degraded tropical forest lands: the potential role of timber plantations. Restoration Ecology, 6(3): 271-279.

Lamb, D., & Gilmour, D. (2003). Rehabilitation and restoration of degraded forests. IUCN, Gland, Switzerland and Cambridge, UK and WWF, Gland, Switzerland, 122pp.

Lamberti, A., Archetti, R., Kramer, M., Paphitis, D., Mosso, C., Di Risio, M. (2005). European experience of low crested structures for coastal management. Coastal Engineering, 52(10-11): 841-866.

Lammerts van Bueren, E.T., Østergard, H., Goldringer, I., Scholten, O. (2008). Plant breeding for organic and sustainable, low-input agriculture: Dealing with genotype-environment interactions.Proceedings of the EUCARPIA symposium of working group organic plant breeding, 7–9 Nov 2007, Wageningen. Euphytica 2008, doi:10.1007/s10681-008-9731-4, pp 1–72

Lamparter, A., Bachmann, J., Goebel, M.O. & Woche, S.K. (2009). Carbon mineralization in soil: Impact of wetting–drying, aggregation and water repellency. Geoderma, 150(3): 324-333.

Laukkonen, J., Blanco, P. K., Lenhart, J., Keiner, M., Cavric, B., & Kinuthia-Njenga, C. (2009). Combining climate change adaptation and mitigation measures at the local level. Habitat International, 33(3): 287-292.

Law B.E. & Harmon M.E. (2011). Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. Carbon Management, 2(1): 73–84.

Lawler, J.J. (2009). Climate change adaptation strategies for resource management and conservation planning. Annals of the New York Academy of Sciences, 1162(1): 79-98.

Lawson, A.R., Greenwood, K.L., & Kelly, K.B. (2009). Irrigation water productivity of winter-growing annuals is higher than perennial forages in northern Victoria. Crop and pasture science, 60(5): 407-419.

Ledgard, S.F. and Steele, K.W. (1992). Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil, 141(1): 137-153.

Ledoux, L. (2003). Wetland values in the context of flood and coastal defence appraisal. Wetland valuation: state of the art and opportunities for further development. In Proceedings of a Workshop Organised for the Environment Agency by Environmental Futures Ltd. and CSERGE. Bristol: Environment Agency (pp. 5-17).

Lee, M. (2001). Coastal defence and the Habitats Directive: predictions of habitat change in England and Wales. Geographical Journal, 167(1): 39-56.

Lehner, B., Liermann, R.C., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M., Frenken, K., Magome, J., Nilsson, C., Robertson, J.C., Rödel, R., Sindorf, N. & Wisser, D. (2011). Global Reservoir and Dam (GRanD) database. Technical documentation. Version 1.1. Available: http://www.gwsp.org/fileadmin/downloads/GRanD_Technical_Documentation_v1_1.pdf Accessed: June 2012.

Lehuger, S., Gabrielle, B., Laville, P., Lamboni, M., Loubet, B., & Cellier, P. (2011). Predicting and mitigating the net greenhouse gas emissions of crop rotations in Western Europe. Agricultural and Forest Meteorology, 151(12): 1654-1671.

Leifeld, J., and Fuhrer, J. (2005). Greenhouse gas emissions from Swiss agriculture since 1990: implications for environmental policies to mitigate global warming. Environmental Science & Policy, 8(4): 410-417.

Leip, A., Britz, W., Weiss, F., & de Vries, W. (2011). Farm, land, and soil nitrogen budgets for agriculture in Europe calculated with CAPRI. Environmental Pollution, 159(11): 3243-3253.

Leskovar, V.Ž., & Premrov, M. (2012a). Design Approach for the Optimal Model of an Energy-Efficient Timber Building with Enlarged Glazing Surface on the South Façade. Journal of Asian Architecture and Building Engineering, 11(1): 71-78.

Leskovar, V.Ž., & Premrov, M. (2012b). Influence of glazing size on energy efficiency of timber-frame buildings. Construction and Building Materials, 30: 92-99.

Leuzinger, S., Vogt, R., & Körner, C. (2010). Tree surface temperature in an urban environment. Agricultural and Forest Meteorology, 150(1): 56-62.

Li, F.M., Guo, A.H., Wei, H. (1999). Effects of plastic film mulch on yield of spring wheat. Field Crops Research, 63(1): 79-86.

Li, F.M., Song, Q.H., Liu, H.S., Li, F.R., & Liu, X.L. (2001). Effects of pre-sowing irrigation and phosphorus application on water use and yield of spring wheat under semi-arid conditions. Agricultural Water Management, 49(3): 173-183.

Li, F.M., Wang, P., Wang, J., Xu, J.Z. (2004). Effects of irrigation before sowing and plastic film mulching on yield and water uptake of spring wheat in semiarid Loess Plateau of China. Agricultural Water Management, 67(2): 77-88.

Li, H., Qiu, J.J., Wang, L.G., Ren, T.Z. (2012). The characterization of greenhouse gases fluxes from croplands of China and mitigation technologies. Ecology and Environmental Sciences, 21(1):159-165.

Li, L.C., Zhang, C.J., Han, H.Z. (2007). Relevant problems in the development of water-saving irrigation. Shandong Water Resources, 6:37-38

Li, Z., Boyle, F., & Reynolds, A. (2010). Rainwater harvesting and greywater treatment systems for domestic application in Ireland. Desalination, 260(1): 1-8.

Li, Z., Liu, W.Z., Zhang, X.C., Zheng, F.L. (2009). The impacts of future climate change on water resources in the Heihe Watershed on the Loess Plateau. Acta Ecologica Sinica 29, 3456-3464. (In Chinese with English abstract)

Liikanen, A., Silvennionen, H, Karvo, A., Rantakokko, P., Martikainen, P.J. (2009). Methane and nitrous oxide fluxes in two coastal wetlands in the northeastern Gulf of Bothnia, Baltic Sea. Boreal Environment Research, 14(3): 351-368.

Lima, I.B., Ramos, F.M., Bambace, L.A., & Rosa, R.R. (2008). Methane emissions from large dams as renewable energy resources: a developing nation perspective. Mitigation and Adaptation Strategies for Global Change, 13(2): 193-206.

Lin, E. (1996). Agricultural vulnerability and adaptation to global warming in China. Water, Air and Soil Pollution, 92(1-2): 63-73.

Lindner, M., Gonzalo, J.G., Kolström, M., Green, T., Reguera, R., Maroschek, M., Seidl, R., Lexer, M.J., Netherer, S., Schopf, A., Kremer, A., Delzon, S., Barbati, A., Machetti, M., Corona, P. (2008). Impacts of climate change on European forests and options for adaptation. Report to the European Commission Directorate-General for Agriculture and Rural Development, 173pp.

Liu, C.A., Jin, S.L., Zhou, L.M., Jia, Y., Li, F.M., Xiong, Y.C., Li, X.G. (2009). Effects of plastic film mulch and tillage on maize productivity and soil parameters. European Journal of Agronomy, 31(4): 241-249.

Liu, C.L., Golding, D., Gong, G. (2008). Farmers' coping response to the low flows in the lower Yellow River: A case study of temporal dimensions of vulnerability. Global Environmental Change, 18(4): 543-553.

Liu, C.M., Li, L.J. (2002). Measures and Problems of Water Resources in North China. Science Technology and Industry, 2(2): 44-50.

Liu, H.J., Kang, Y.H., Liu, S.P. (2003). Regulation of field environmental condition by sprinkler irrigation and its effect on water use efficiency of winter wheat. Trans Chin Soc Agric Eng, 19: 46-51.

Liu, J.G., Wiberg, D., Zehnder, A.J.B., Yang, H. (2007). Modeling the role of irrigation in winter wheat yield, crop water productivity, and production in China. Irrig Sci, 26: 21-23.

Liu, X.Q., Zhao, J.B., Yu, X.F. (2006). Study on the climatic warming-drying trend in the Loess Plateau and the countermeasures. Arid Zone Research, 23(4):627-631.

Loomis, M.J., and Craft, C.B. (2010). Carbon Sequestration and Nutrient (Nitrogen, Phosphorus) Accumulation in River-Dominated Tidal Marshes, Georgia, USA. Soil Sci. Soc. Am. J., 74(3): 1028-1036.

Lopes-Alves, M.F., Ferreira, J.C. (2004). 10 years after the Rio Summit: the assessment of Portuguese coastal zone planning system. Journal of Coastal Research, 39, 1311–1315

Loss, S.R., Terwilliger, L.A., & Peterson, A.C. (2011). Assisted colonization: Integrating conservation strategies in the face of climate change. Biological conservation, 144(1): 92-100.

Louters, T., van den Berg, J.H., Mulder, J.P.M, (1998). Geomorphological Changes of the Oosterschelde Tidal System during and after the Implementation of the Delta Project. Journal of Coastal Research 14(3): 1134-1151.

Lugato, E., and Berti, A. (2008). Potential carbon sequestration in a cultivated soil under different climate change scenarios: A modelling approach for evaluating promising management practices in north-east Italy. Agriculture, Ecosystems & Environment, 128(1–2): 97-103.

Luisetti, T., Turner, K.R., Bateman, I.J., Morse-Jones, S., Adams, C., Fonseca, L., (2011). Coastal and marine ecosystem services valuation for policy and management: Managed realignment case studies in England. Ocean & Coastal Management, 54(3): 212-224.

Luisetti, T., Turner, K.R., Hadley, D., Morse-Jones, S., (2010). Coastal and marine ecosystem services valuation for policy and management. CSERGE working paper EDM: 10-04

Lundholm, J., MacIvor, J.S., MacDougall, Z., & Ranalli, M. (2010). Plant species and functional group combinations affect green roof ecosystem functions. Plos One, 5(3): e9677.

Luo, J., De Klein, C.A.M., Ledgard, S.F., & Saggar, S. (2010). Management options to reduce nitrous oxide emissions from intensively grazed pastures: a review. Agriculture, Ecosystems & Environment, 136(3): 282-291.

Lutsey, N., & Sperling, D. (2008). America's bottom-up climate change mitigation policy. Energy Policy, 36(2): 673-685.

Ma, L.J., Feng, M. (2006). The development of water saving irrigation is a way out of agriculture. Water Conservancy Science and Technology and Economy, 12(6):394.

Mackey, C.W., Lee, X., & Smith, R.B. (2012). Remotely sensing the cooling effects of city scale efforts to reduce urban heat island. Building and Environment, 49: 348-358.

MacLeod, C.L., Scrimshaw, M.D., Emmerson, R.H.C., Change, Y.-H., Lester, J.N., (1999). Geochemical Changes in Metal and Nutrient Loading at Orplands Farm Managed Retreat Site, Essex, UK (April 1995-1997). Marine Pollution Bulletin, 38(12): 1115-1125.

Maddrell, R. J. (1996). Managed coastal retreat, reducing flood risks and protection costs, Dungeness Nuclear Power Station, UK. Coastal Engineering 28(1-4): 1-15.

Mader, T.L., Davis, M.S., & Brown-Brandl, T. (2006). Environmental factors influencing heat stress in feedlot cattle. Journal of Animal Science, 84(3): 712-719.

Magurran, A.E., Baillie, S.R., Buckland, S.T., Dick, J.M., Elston, D.A., Scott, M.E., Smith, R.I., Somerfield, P.J., Watt, A.D. (2010). Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology & Evolution* 25(10): 574-582.

Maiorano, L., Falcucci, A., Zimmermann, N.E., Psomas, A., Pottier, J., Baisero, D., Rondinini, C., Guisan, A., Boitani, L. (2011). The future of terrestrial mammals in the Mediterranean basin under climate change. Philosophical Transactions of the Royal Society B: Biological Sciences, 366(1578): 2681-2692.

Malgeryd, J. (1998). Technical measures to reduce ammonia losses after spreading of animal manure. Nutrient Cycling in Agroecosystems, 51(1): 51-57.

Mander, L., Cutts, N.D., Allen, J., Mazik, K., (2007). Assessing the development of newly created habitat for wintering estuarine birds. Estuarine, Coastal and Shelf Science, 75(1-2): 163-174.

Manderscheid, R., & Weigel, H.J. (1995). Do increasing atmospheric CO2 concentrations contribute to yield increases of German crops? Journal of Agronomy and Crop Science, 175: 73-82.

Mangin, S., and Valdes, S. (2005). Conserving coastal habitats one partner at a time. Oceans, 2005. Proceedings of MTS/IEEE, 3: 2035-2037.

Marcus, L. (2000). Restoring tidal wetlands at Sonoma Baylands, San Francisco Bay, California. Ecological Engineering, 15(3-4): 373-383.

Marquiegui, M.A. & Aguirrezabalaga, F. (2009). Colonization process by macrobenthic infauna after a managed coastal realignment in the Bidasoa estuary (Bay of Biscay, NE Atlantic). Estuarine, Coastal and Shelf Science, 84(4): 598-604.

Martin, D., Bertasi, F., Colangelo, M.A., de Vries, M., Frost, M., Hawkins, S.J., MacPherson, E., Moschella, P.S., Satta, P.M., Thompson, R.C., Ceccherelli, V.U. (2005). Ecological impact of coastal defence structures on sediment and mobile fauna: Evaluating and forecasting consequences of unavoidable modifications of native habitats. Coastal Engineering, 52(10-11): 1027-1051.

Massé, D.I., Masse, L., Claveau, S., Benchaar, C., & Thomas, O. (2008). Methane emissions from manure storages. Transactions of the ASABE, 51(5): 1775-1781.

Mauchamp, A., Chauvelon, P., Grillas, P. (2002). Restoration of floodplain wetlands: Opening polders along a coastal river in Mediterranean France, Vistre marshes. Ecological Engineering, 18: 619-632.

Mavrogianni, A., & Mumovic, D. (2010). On the use of windcatchers in schools: climate change, occupancy patterns, and adaptation strategies. Indoor and Built Environment, 19(3): 340-354.

Mavrogianni, A., Wilkinson, P., Davies, M., Biddulph, P., & Oikonomou, E. (2012). Building characteristics as determinants of propensity to high indoor summer temperatures in London dwellings. Building and Environment, 55: 117-130.

Mazik, K., Musk, W., Dawes, O., Solyanko, K., Brown, S., Mander, L., Elliott, M., (2010). Managed realignment as compensation for the loss of intertidal mudflat: A short term solution to a long term problem? Estuarine, Coastal and Shelf Science, 90(1): 11-20.

Mazik, K., Smith, J.E., Leighton, A., Elliott, M. (2007). Physical and biological development of a newly breached managed realignment site, Humber estuary, UK. Marine Pollution Bulletin, 55(10-12): 564-578.

McAneney, K.J., Arrue, J.L. (1993). A wheat-fallow rotation in northeastern Spain: water balance-yield considerations. Agronomie 13: 481-490.

McCarl, B.A., & Schneider, U.A. (2001). The cost of greenhouse gas mitigation in US agriculture and forestry. Science, 294(21): 2481-82.McCulloch, J. S. G. & Robinson, M. 1993. History of forest hydrology. Journal of Hydrology, 150, 189-216.

McEvoy, J. & Wilder, M. (2012). Discourse and desalination: Potential impacts of proposed climate change adaptation interventions in the Arizona–Sonora border region. Global Environmental Change, 22: 353-363.

McFadden, L. (2008). Exploring the challenges of integrated coastal zone management and reflecting on contributions to 'integration' from geographical thought. Geographical Journal, 174(4): 299-314.

McKenney, D., Pedlar, J., & O'Neill, G. (2009). Climate change and forest seed zones: past trends, future prospects and challenges to ponder. The Forestry Chronicle, 85(2): 258-266.

McLeod E. and Salm R. V., 2006. Managing Mangroves for Resilience to Climate Change. IUCN, Gland, Switzerland. 64pp.

McPherson, E.G., & Rowntree, R.A. (1993). Energy conservation potential of urban tree planting. Journal of Arboriculture, 19: 321-321.

McPherson, E.G., & Simpson, J.R. (2003). Potential energy savings in buildings by an urban tree planting programme in California. Urban Forestry & Urban Greening, 2(2): 73-86.

McPherson, G., Simpson, J.R., Peper, P.J., Maco, S.E., Xiao, W. (2005). Municipal Forest Benefits and Costs in Five US Cities. Journal of Forestry, 103(8): 411-416.

MEA, (2005). Ecosystems and Human Well-being: Current State and Trends. Millennium Ecosystem Assessment, Washington DC: Island Press.

Melero, S., López-Garrido, R., Murillo, J.M., & Moreno, F. (2009). Conservation tillage: Short-and long-term effects on soil carbon fractions and enzymatic activities under Mediterranean conditions. Soil and Tillage Research, 104(2): 292-298.

Melia, S., Parkhurst, G., Barton, H. (2011). The paradox of intensification. Transport Policy, 18(1): 46-52.

Mendum, R., & Glenna, L.L. (2010). Socioeconomic Obstacles to Establishing a Participatory Plant Breeding Program for Organic Growers in the United States. Sustainability, 2: 73-91.

MENR-AZ (2010). United Nations Development Programme (UNDP) ,Global Environment Facility (GEF), 2010. Azerbaijan. Azerbaijan second national communication to the United Nations Framework Convention on Climate Change. Ministry of Ecology and Natural Resources, Submission date: 1 December 2010, Baku.

Mentens, J., Raes, D., & Hermy, M. (2006). Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? Landscape and urban planning, 77(3): 217-226.

MEPNR-GE (2009). Ministry of Environment Protection and Natural Resources, 2009. Georgia. Georgia's second national communication to the UNFCCC. Submission date: 2 October 2009, Tbilisi.

Merse, C.L., Buckley, G.L., Boone, C.G. (2009). Street trees and urban renewal: A Baltimore case study. Geographical Bulletin - Gamma Theta Upsilon, 50(2): 65-81.

Meyer-Aurich, A., Olesen, J. E., Prochnow, A., & Brunsch, R. (2012). Greenhouse gas mitigation with scarce land: The potential contribution of increased nitrogen input. Mitigation and Adaptation Strategies for Global Change: 1-12.

Meza, F.J., Wilks, D.S. (2003). Value of operational forecasts of seasonal average sea surface temperature anomalies for selected rain-fed agricultural locations of Chile. Agric. For. Meteorol. 116: 137-158.

Ministry of Water Resource (2012). Bulletin of Water Resource in 2010. http://www.mwr.gov.cn/zwzc/hygb/szygb/qgszygb/201204/t20120426_319624.html

Mitch, W.J., Gosselink, J.G. (2000). Wetlands. 3rd Ed. John Wiley & Sons, New York, NY.

MNP-AM (2010). Ministry of Nature Protection, 2010. Armenia. Second national communication under the United Nations Framework Convention on Climate Change. Submission date: 7 September 2010, Yerevan.

Möller, I., Spencer, T., French, J.R., Leggett, D.J., Dixon, M., (2001). The sea-defence value of salt marshes: field evidence from north Norfolk. Water and Environment Journal, 15(2): 109-116.

Monahan, J., & Powell, J.C. (2011). A comparison of the energy and carbon implications of new systems of energy provision in new build housing in the UK. Energy Policy, 39(1): 290-298.

Morgounov, A., Zykin, V., Belan, I., Roseeva, L., Zelenskiy, Y., Gomez-Becerra, H.F., Budak, H., Bekes, F. (2010). Genetic gains for grain yield in high latitude spring wheat grown in Western Siberia in 1900–2008. Field Crops Research, 117(1): 101-112.

Moriondo, M., Bindi, M., Kundzewicz, Z., Szwed, M., Chorynski, A., Matczak, P., Radziejewski, M., McEvoy, D., Wreford, A. (2010) Impact and adaptation opportunities for European agriculture in response to climatic change and variability. Mitigation and Adaptation Strategies for Global Change, 15(7): 657-679.

Morris, J.T., and Jensen, A. (1998). The carbon balance of grazed and non-grazed Spartina anglica saltmarshes at Skallingen, Denmark. Journal of Ecology, 86(2): 229-242.

Morueta-Holme, N., Fløjgaard, C., & Svenning, J.C. (2010). Climate change risks and conservation implications for a threatened small-range mammal species. PLoS One, 5(4): e10360.

Moschella, P.S., Abbiati, M., Åberg, P., Airoldi, L., Anderson, J.M., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M., Gacia, E., Granhag, L., Jonsson, P.R., Satta, M.P., Sundelöf, A., Thompson, R.C., Hawkins, S.J., (2005). Low-crested coastal defence structures as artificial habitats for marine life: Using ecological criteria in design. Coastal Engineering, 52(10-11): 1053-1071.

Moseman-Valtierra, S., Gonazalez, R., Kroeger, K.D., Tang, J., Chao, W.C., Crusuis, J., Bratton, J., Green, A., Shelton, J., (2011). Short-term nitrogen additions can shift a coastal wetland from a sink to a source of N2O. Atmospheric Environment, 45(26): 4390-4397.

Moser, S.C. (2009). Governance and the art of overcoming barriers to adaptation. Magazine of the International Human Dimensions Programme on Global Environmental Change, 31.

Mueller, J.M., & Hellmann, J.J. (2008). An assessment of invasion risk from assisted migration. Conservation Biology, 22(3): 562-567.

Munoz, A., Lopez-Pineiro, A., & Ramirez, M. (2007). Soil quality attributes of conservation management regimes in a semi-arid region of south western Spain. Soil and Tillage Research, 95(1): 255-265.

Mushtaq, S., & Moghaddasi, M. (2011). Evaluating the potentials of deficit irrigation as an adaptive response to climate change and environmental demand. Environmental Science & Policy, 14(8): 1139-1150.

Mwenge Kahinda, J., Taigbenu, A.E. & Boroto, R.J. (2010). Domestic rainwater harvesting as an adaptation measure to climate change in South Africa. Physics and Chemistry of the Earth, Parts A/B/C, 35: 742-751.

Nabuurs, G.J., Pussinen, A., Brusselen, J., Schelhaas, M.J. (2007). Future harvesting pressure on European forests. European Journal of Forest Research, 126(3): 391-400

Nahm, K.H. (2007). Efficient phosphorus utilization in poultry feeding to lessen the environmental impact of excreta. World's Poultry Science Journal, 63(4): 625-654.
Nardone, A., Ronchi, B., Lacetera, N., Ranieri, M.S., Bernabucci, U. (2010). Effects of climate changes on animal production and sustainability of livestock systems. Livestock Science, 130: 57–69.

Natura (2000). European Comission DG ENV Nature Newsletter, 11: 12pp. Available: http://ec.europa.eu/environment/nature/info/pubs/docs/nat2000newsl/nat11_en.pdf Accessed: July 2013.

Natura (2007). Biodiversity and climate change: the role of the Natura 2000 network. European Comission DG ENV Nature Newsletter, 22: 16pp. Available: http://ec.europa.eu/environment/nature/info/pubs/docs/nat2000newsl/nat22_en.pdf Accessed: July 2013.

Natural England (2010). Making space for wildlife in a changing environment (NE263): information for local authorities in the South East. Natural England, NE263: 8pp. Available: http://publications.naturalengland.org.uk/publication/47002 Accessed: July 2013.

Nelson, W.G. (1993). Beach restoration in the southeastern US: Environmental effects and biological monitoring. Ocean & Coastal Management, 19(2): 157-182.

Neufeldt, H., and Schäfer, M. (2008). Mitigation strategies for greenhouse gas emissions from agriculture using a regional economic-ecosystem model. Agriculture, Ecosystems & Environment, 123(4): 305-316.

Newbould, P., Holding, A.J., & Lake, J.V. (1982). Biological Nitrogen Fixation in Upland and Marginal Areas of the UK [and Discussion]. Philosophical Transactions of the Royal Society of London. B, Biological Sciences, 296(1082): 405-417.

Newton, I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. Ibis, 146(4): 579-600.

Nienaber, J.A. and Hahn, G.L. (2007). Livestock production system management responses to thermal challenges. International Journal of Biometeorology, 52(2): 149-157.

Noordwijk-Puijk, K., Beeftink, W.G., Hogeweg, P., (1979). Vegetation development on salt-marsh flats after disappearance of the tidal factor. Plant Ecology, 39(1): 1-13.

Nordstrom, K.F., Lampe, R., Jackson, N.L. (2007). Increasing the dynamism of coastal landforms by modifying shore protection methods: examples from the eastern German Baltic Sea Coast. Environmental Conservation, 34(3): 205-214.

Norton, D.A. (1998). Indigenous biodiversity conservation and plantation forestry: options for the future. New Zealand Forestry, 43: 34-39.

Nowak, D.J. (1994). Atmospheric carbon dioxide reduction by Chicago's urban forest. USDA Forest Service General Technical Report NE-186, Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. D.J.N.a.R.A.R.E.G. McPherson: 83–94.

Nowak, D.J. and Crane, D.E. (2002). Carbon storage and sequestration by urban trees in the USA. Environmental Pollution, 116(3): 381-389.

O'Rourke, S.M., Foy, R.H., Watson, C.J., Gordon, A., & Doody, D. (2012). Assessment of co-blending water treatment residual with dairy manure to reduce phosphorus concentrations in run-off in Northern Ireland. Soil Use and Management, 28(2): 157-166.

Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R.R., Doshi, H., Dunnett, N., Gaffin, S., Köhler, M., Lui, K.K.Y., Rowe, B. (2007). Green roofs as urban ecosystems: ecological structures, functions, and services. BioScience, 57(10): 823-833.

O'Brien, K.L., & Leichenko, R.M. (2000). Double exposure: assessing the impacts of climate change within the context of economic globalization. Global environmental change, 10(3): 221-232.

OECD (2010). Sustainable management of water resources in agriculture. OECD, 122pp.

Olesen, J.E. and Bindi, M. (2002). Consequences of climate change for European agricultural productivity, land use and policy. European Journal of Agronomy, 16: 239-262.

Olesen, J.E., Jensen, T., & Petersen, J. (2000). Sensitivity of field-scale winter wheat production in Denmark to climate variability and climate change. Climate Research, 15(3): 221-238.

Olesen, J.E., Rubæk, G.H., Heidmann, T., Hansen, S., & Børgensen, C.D. (2004). Effect of climate change on greenhouse gas emissions from arable crop rotations. Nutrient Cycling in Agroecosystems, 70(2): 147-160.

Olsen, Y.S., Dausse, A., Garbutt, A., Ford, H., Thomas, D.N., Jones, D.L. (2011). Cattle grazing drives nitrogen and carbon cycling in a temperate salt marsh. Soil Biology and Biochemistry, 43(3): 531-541.

ONERC (2007). Strategie nationale d'adaptation au changement climatique (National Adaptation Strategy to Climate Change). Observatoire National sur les Effectes du Rechauffement Climatique (ONERC), Ministry of Ecology and Sustainable Development, 97pp.

Onmura, S., Matsumoto, M., & Hokoi, S. (2001). Study on evaporative cooling effect of roof lawn gardens. Energy and Buildings, 33(7): 653-666.

Onmura, S., Matsumoto, M., Hokoi, S., (2001). Study on evaporative cooling effect of roof lawn gardens. Energy and Buildings, 33(7): 653-666.

Oorts, K., Merckx, R., Gréhan, E., Labreuche, J., & Nicolardot, B. (2007). Determinants of annual fluxes of CO2 and N2O in long-term no-tillage and conventional tillage systems in northern France. Soil and Tillage Research, 95(1): 133-148.

Ortigosa, L. & García-Ruiz, J.M. (1995). Geomorphological consequences of afforestation at a basin scale, an example from the Central Pyrenees. Physics and Chemistry of the Earth, 20: 345-349.

Ortiz, R., Sayre, K.D., Govaerts, B., Gupta, R., Subbarao, G.V., Ban, T., Hodson, D., Dixon, J.M., Ortiz-Monasterio, J.I., Reynolds, M. (2008). Climate change: Can wheat beat the heat? Agriculture, Ecosystems & Environment, 126(1): 46-58.

Osada, T., Takada, R., & Shinzato, I. (2011). Potential reduction of greenhouse gas emission from swine manure by using a low-protein diet supplemented with synthetic amino acids. Animal Feed Science and Technology, 166: 562-574.

Ostle, N.J., Levy, P.E., Evans, C.D. & Smith, P. (2009). UK land use and soil carbon sequestration. Land Use Policy, 26: S274-S283.

Ottelé, M., Perini, K., Fraaij, A.L.A., Haas, E.M., & Raiteri, R. (2011). Comparative life cycle analysis for green façades and living wall systems. Energy and Buildings, 43(12): 3419-3429.

Pan, G.X., Gao, M., Hu, G.H., Wei, Q.P., Yang, X.G., Zhang,W.Z., Zhou, G.S., Zou, J.W. (2011). Issues and challenges on mitigation of climate change impacts on China's future agriculture. Journal of Agro-Environment Science, 30(9):1707-1712.

Parrott, A., & Burningham, H. (2008). Opportunities of, and constraints to, the use of intertidal agri-environment schemes for sustainable coastal defence: A case study of the Blackwater Estuary, southeast England. Ocean & Coastal Management, 51(4): 352-367.

Parry, M., Arnell, N., Berry, P., Dodman, D., Fankhauser, S., Hope, C., Kovats, S.,R.,N., Satterthwaite, D., Tiffin, R. & Wheeler, T. (2009). Assessing the costs of adaptation to climate change. A review of the UNFCCC and other recent estimates, London, International Institute for Environment and Development and Grantham Institute of Climate Change, London, 116pp.

Parry, M.A.J., Flexas, J. and Medrano, H. (2005). Prospects for crop production under drought: research priorities and future directions. Annals of Applied Biology, 147 (3): 211-226.

Parry, M.L., Carter, T.R., & Konijn, N.T. (1988). *The impact of climatic variations on agriculture: assessments in semi-arid regions, Vol. 2*. Kluwer Academic, 776pp.

Pataki, D.E., Carriero, M.M., Cherrier, J., Grulke, N.E., Jennings, V., Pincetl, S., Pouyat, R.V., Whitlow, T.H., Zipperer, W.C. (2011). Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. Frontiers in Ecology and the Environment, 9(1): 27-36.

Patosaari, P. (2007). Forests and Climate Change: Mitigation and Adaptation through Sustainable Forest Management. Remarks presented to the 60th annual DPI/NGO conference. Climate change: How it impacts us all. UN, New York, 6: 7pp.

Pauleit, S., Jones, N., Garcia-Martin, G., Garcia-Valdecantos, J.L., Rivière, L.M., Vidal-Beaudet, L., Monique Bodson, M., Thomas B. Randrup, T.B. (2002). Tree establishment practice in towns and cities – Results from a European survey. Urban Forestry & Urban Greening, 1(2): 83-96.

Paustian, K.A.O.J.H., Andrén, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M., Woomer, P.L. (1997). Agricultural soils as a sink to mitigate CO2 emissions. Soil use and management 13(s4): 230-244.

Pearson, R.G., & Dawson, T.P. (2005). Long-distance plant dispersal and habitat fragmentation: identifying conservation targets for spatial landscape planning under climate change. Biological Conservation, 123(3): 389-401.

PELCOM (2000). Pan-European land use and land cover monitoring-1km. Available: http://cgi.girs.wageningen-ur.nl/cgi/projects/eu/PELCOM/public/index.htm Accessed: June 2013.

Pelini, S.L., Dzurisin, J.D., Prior, K.M., Williams, C.M., Marsico, T.D., Sinclair, B.J., & Hellmann, J.J. (2009). Translocation experiments with butterflies reveal limits to enhancement of poleward populations under climate change. Proceedings of the National Academy of Sciences, 106(27): 11160-11165.

Peltonen-Sainio, P., Hakala, K., & Jauhiainen, L. (2011). Climate-induced overwintering challenges for wheat and rye in northern agriculture. Acta Agriculturae Scandinavica Section B–Soil and Plant Science, 61(1): 75-83.

Peltonen-Sainio, P., Jauhiainen, L., Hakala, K., & Ojanen, H. (2009). Climate change and prolongation of growing season: changes in regional potential for field crop production in Finland. Agricultural and Food Science, 18(3-4): 171-190.

Peoples, M.B., Herridge, D.F., & Ladha, J.K. (1995). Biological nitrogen fixation: an efficient source of nitrogen for sustainable agricultural production? Plant and soil, 174(1-2): 3-28.

Perini, K., Ottelé, M., Haas, E. M., & Raiteri, R. (2011). Greening the building envelope, façade greening and living wall systems. Open Journal of Ecology, 1(1): 1-8.

Peterson, C.H. and M.J. Bishop (2005). Assessing the Environmental Impacts of Beach Nourishment. BioScience, 55(10): 887-896.

Peterson, C.H., Bishop, M.J., Johnson, G.A., D'Anna, L.M., Manning, L.M. (2006). Exploiting beach filling as an unaffordable experiment: Benthic intertidal impacts propagating upwards to shorebirds. Journal of Experimental Marine Biology and Ecology, 338(2): 205-221.

Pethick, J. (2002). Estuarine and tidal wetland restoration in the United Kingdom: Policy versus practice. Restoration Ecology, 10(3): 431-437.

Pimpão, C.T., Zampronio, A.R., & Silva de Assis, H.C. (2007). Effects of deltamethrin on hematological parameters and enzymatic activity in Ancistrus multispinis (Pisces, Teleostei). Pesticide Biochemistry and Physiology, 88(2): 122-127.

Pizzaro, R. (2009). Urban form and climate change: Towards appropriate development patterns to mitigate and adapt to global warming. In Planning for climate change: Strategies for mitigation and adaptation, (Eds) Davoudi, S., Crawford, J., Mehmood, A. 33–45. London: Earthscan.

Plantinga, A.J., Wu, J.-J. (2003). Co-Benefits from Carbon Sequestration in Forests: Evaluating Reductions in Agricultural Externalities from an Afforestation Policy in Wisconsin. Land Economics, 79(1): 74-85.

PNACC, 2006. Plan Nacional de Adaptacional Cambio Climatico. Primer Programa de Trabajo (National Plan for Adaptation to Climate Change. First Working Programme) Spanish Office for Climate Change, Ministry of the Environment.

Pontee, N. I. (2007). Realignment in low-lying coastal areas: UK experiences. Proceedings of the Institution of Civil Engineers: Maritime Engineering, 160(4): 155-166.

Pontee, N.I., Hull, S.C., Moore, J.R. (2006). Banked realignment: a case study from the Humber Estuary, UK Proceedings of the Institution of Civil Engineers - Engineering Sustainability, 159(3): 99-108.

Powlson, D.S., Bhogal, A., Chambers, B.J., Coleman, K., Macdonald, A.J., Goulding, K.W.T., & Whitmore, A.P. (2012). The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. Agriculture, Ecosystems & Environment, 146(1): 23-33.

Pöyry, J. and Toivonen, H. (2005). Climate change adaptation and biological diversity. FINADAPT Working Paper 3, Finnish Environment Institute Mimeographs 333, Helsinki, 46pp.

Prach, K., Pyšek, P., & Jarošík, V. (2007). Climate and pH as determinants of vegetation succession in Central European man-made habitats. Journal of Vegetation Science, 18(5): 701-710.

Primavera J. H., Esteban J. M. A., 2008. A review of mangrove rehabilitation in the Philippines: successes, failures and future prospects. Wetlands Ecology and Management, 16(5): 345-358.

Purkey, D.R., Joyce, B., Vicuna, S., Hanemann, M. W., Dale, L.L., Yates, D., & Dracup, J.A. (2008). Robust analysis of future climate change impacts on water for agriculture and other sectors: a case study in the Sacramento Valley. Climatic Change, 87(1): 109-122.

Qu, Z.H., Han, Y.Q., Tang, C.M., Geng, G.R., Wu, Z.F. (2011). Survey and analysis of agricultural water conservation status in Northwest Hebei. Journal of Anhui Agri Sci, 39(35): 21932-21934.

Quétier, F., & Lavorel, S. (2011). Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. Biological Conservation, 144(12): 2991-2999.

Quevauviller, P. (2011). Adapting to climate change: reducing water-related risks in Europe – EU policy and research considerations. Environmental Science & Policy, 14: 722-729.

Quiggin, J., Adamson, D., Chambers, S., & Schrobback, P. (2010). Climate change, uncertainty, and adaptation: The case of irrigated agriculture in the Murray–Darling Basin in Australia. Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie, 58(4): 531-554.

Ragauskas, A.J., Williams, C.K., Davison, B.H., Britovsek, G., Cairney, J., Eckert, C.A., Frederick Jr., W.J., Hallett, J.P., Leak, D.J., Liotta, C.L., Mielenz, J.R., Murphy, R., Templer, R., Tschaplinski, T. (2006). The path forward for biofuels and biomaterials. Science, 311(5760): 484-489.

Raulund-Rasmussen, K., De Jong, J., Humphrey, J.W., Smith M., Ravn, H.P., Katzensteiner, K., Klimo, E., Szukics, U., Delaney, C., Hansen, K., Stupak, I., Ring, E., Gundersen, P. and Loustau, D. (2011). Papers on impacts of forest management on environmental services.

Reading, C.J., Garbutt, R.A., Watts, C.W., Rothery, P.,Turk, A.,Yates, M., Boffey, C., Saunders, J., & Wolters, M. (2008). Managed realignment at Tollesbury: R&D Technical Report FD 1922/TR. Joint Defra/EA Flood and Coastal Erosion Risk Management R&D Programme, 126pp.

Reidsma, P., Lansink, A.O., & Ewert, F. (2009). Economic impacts of climatic variability and subsidies on European agriculture and observed adaptation strategies. Mitigation and Adaptation Strategies for Global Change, 14(1): 35-59.

Reilly, J. (1995). Climate Change and Agriculture: Recent Findings and Issues. American Journal of Agricultural Economics, 77: 727-733

Reise, K. (1998). Coastal change in a tidal backbarrier basin of the northern Wadden Sea: Are tidal flats fading away? Marine Biodiversity, 29(1): 121-127.

Reise, K. (2005). Coast of change: habitat loss and transformations in the Wadden Sea. Helgoland Marine Research, 59(1): 9-21.

Ricciardi, A., & Simberloff, D. (2009). Assisted colonization is not a viable conservation strategy. Trends in Ecology & Evolution, 24(5): 248-253.

Richards, R.A. (2006). Physiological traits used in the breeding of new cultivars for water-scarce environments. Agricultural Water Management, 80(1): 197-211.

Richards, RA. (1996). Defining selection criteria to improve yield under drought. Plant Growth Regulation, 20: 57-166.

Robinson, M., Cognard-Plancq, A.L., Cosandey, C., David, J., Durand, P., Führer, H.W., Hall, R., Hendriques, M. O., Marc, V., McCarthy, R., McDonnell, M., Martin, C., Nisbet, T., O'Dea, P., Rodgers, M. & Zollner, A. (2003). Studies of the impact of forests on peak flows and baseflows: a European perspective. Forest Ecology and Management, 186: 85-97.

Robledo C., Kanninen M., Pedroni L. (2005). Tropical forests and adaptation to climate change: In search of synergies. CIFOR, 195pp.

Robledo, C. and Forner, C. (2005). Adaptation of forest ecosystems and the forest sector to climate change. FAO, Forests and Climate Change Working Paper, 2: 88pp

Rodriguez, R.J., Henson, J., Van Volkenburgh, E., Hoy, M., Wright, L., Beckwith, F., Kim, Y.-O., Redman, R.S. (2008). Stress tolerance in plants via habitat-adapted symbiosis. The ISME journal 2(4): 404-416.

Rohde, S., Hostmann, M., Peter, A. & Ewald, K.C. (2006). Room for rivers: An integrative search strategy for floodplain restoration. Landscape and Urban Planning, 78: 50-70.

Roman, C.T., Raposa, K.B., Adamowicz, S.C., James-Pirri, M.-J., Catena, J.G. (2002). Quantifying vegetation and nekton response to tidal restoration of a New England salt marsh. Restoration Ecology, 10(3): 450-460.

Rose, N.A., & Burton, P.J. (2009). Using bioclimatic envelopes to identify temporal corridors in support of conservation planning in a changing climate. Forest Ecology and Management, 258: S64-S74.

Rosenzweig, C., & Tubiello, F.N. (2007). Adaptation and mitigation strategies in agriculture: an analysis of potential synergies. Mitigation and Adaptation Strategies for Global Change, 12(5): 855-873.

Rosenzweig, C., Strzepek, K.M., Major, D.C., Iglesias, A., Yates, D.N., McCluskey, A., Hillel, D. (2004). Water resources for agriculture in a changing climate: International case studies. Global Environmental Change, 14(4): 345-360.

Rotherham, I.D. (2010). Thoughts on the Politics and Economics of Urban Street Trees. Arboricultural Journal, 33(2): 69-75.

Rotman, R., Naylor, L., McDonnell, R., MacNiocaill, C. (2008). Sediment transport on the Freiston Shore managed realignment site: An investigation using environmental magnetism. Geomorphology, 100(3-4): 241-255.

Rowe, D. B. (2011). Green roofs as a means of pollution abatement. Environmental Pollution, 159(8): 2100-2110.

Rozas, L.P., Caldwell, P., Minello, T.J., (2005). The Fishery Value of Salt Marsh Restoration Projects. Journal of Coastal Research, 40: 37-50.

Saeijs, H.L.F. & Stortelder, P.B.M. (1982). The Fishery Value of Salt Marsh Restoration Projects: Converting an estuary to Lake Grevelingen: Environmental review of a coastal engineering project. Environmental Management, 6(5): 377-405.

Saiz-Salinas, J.I., & Urkiaga-Alberdi, J. (1999). Faunal responses to turbidity in a man-modified bay (Bilbao, Spain). Marine Environmental Research, 47(4): 331-347.

Saiz-Salinas, J.I., and Urkiaga-Alberdi, J. (1999). Use of Faunal Indicators for Assessing the Impact of a Port Enlargement near Bilbao (Spain). Environmental Monitoring and Assessment, 56(3): 305-330.

Sánchez-Fernández, D., Bilton, D.T., Abellán, P., Ribera, I., Velasco, J., & Millán, A. (2008). Are the endemic water beetles of the Iberian Peninsula and the Balearic Islands effectively protected? Biological Conservation, 141(6): 1612-1627.

Sanders, O., Goesch, T. and Hughes, N. (2010). Adapting to Water Scarcity. Canberra: Australian Bureau of Agricultural and Resource Economics, Issues and Insights, 10.5: 17pp.

Sandler, R. (2010). The value of species and the ethical foundations of assisted colonization. Conservation Biology, 24(2): 424-431.

Santamouris, M., Gaitani, N., Spanou, A., Saliari, M., Giannopoulou, K., Vasilakopoulou, K., & Kardomateas, T. (2012). Using cool paving materials to improve microclimate of urban areas–Design realization and results of the flisvos project. Building and Environment, 53: 128-136.

Sarker, A., & Erskine, W. (2006). Recent progress in the ancient lentil. Journal of Agricultural Science, 144(1): 19.

Sauer, T., Havlík, P., Schneider, U.A., Schmid, E., Kindermann, G., & Obersteiner, M. (2010). Agriculture and resource availability in a changing world: The role of irrigation. Water Resources Research, 46(6): W06503.

Schekkerman, H., Meininger, P.L., Meire, P.M., (1994). Changes in the waterbird populations of the Oosterschelde (SW Netherlands) as a result of large-scale coastal engineering works. Hydrobiologia, 282-283(1): 509-524.

Scherba, A., Sailor, D.J., Rosenstiel, T.N., & Wamser, C.C. (2011). Modeling impacts of roof reflectivity, integrated photovoltaic panels and green roof systems on sensible heat flux into the urban environment. Building and Environment, 46(12): 2542-2551.

Schmidt, L., Prista, P., Saraiva, T., O'Riordan, T., & Gomes, C. (2012). Adapting governance for coastal change in Portugal. Land Use Policy, 31: 314-325.

Scholz, M. (2006). Best management practice: A sustainable urban drainage system management case study. Water international, 31(3): 310-319.

Scholz, M., Corrigan, N.L., & Yazdi, S.K. (2006). The Glasgow sustainable urban drainage system management project: Case studies (Belvidere hospital and Celtic FC stadium areas). Environmental Engineering Science, 23(6): 908-922.

Schreurs, M. A. (2008). From the bottom up local and subnational climate change politics. The Journal of Environment & Development, 17(4): 343-355.

Schröder, K. (2008). Root space underneath traffic lanes. Arboricultural Journal, 31(1): 33-43.

Schwartz, M.W., Hellmann, J.J., & McLachlan, J.S. (2009). The precautionary principle in managed relocation is misguided advice. Trends in Ecology & Evolution, 24(9): 474.

Searle, G. (2010). Too concentrated? The planned distribution of residential density in SEQ. Australian Planner, 47(3): 135-141.

Secretariat of the Convention on Biological Diversity (2009). connecting biodiversity and climate change mitigation and adaptation: Report of the second ad hoc technical expert group on biodiversity and climate change. Montreal, Technical Series, 41: 126pp.

Sedjo A. R., (2010). Adaptation of Forests to Climate Change: Some Estimates. Discussion Paper, RFF DP 10-06, 54pp.

Sedjo, A.R. (1991). Economic Aspects of Climate, Forests, and Fire: A North American Perspective. Environment International, 17: 2-3.

Selvaraju, R., & Sobbiah, A.R. (2006). *Livelihood Adaptation to Climate Variability and Change in Drought-prone Areas Bangladesh*. FAO, 88pp.

Semadeni-Davies, A., Hernebring, C., Svensson, G., & Gustafsson, L.G. (2008). The impacts of climate change and urbanisation on drainage in Helsingborg, Sweden: Suburban stormwater. Journal of Hydrology, 350(1): 114-125.

Seppala R., Chair P., Buck A., Katila P. (2009). Adaptation of forests and people to climate change - A Global Assessment Report. IUFRO World Series Volume 22, Helsinki, 224pp.

Shepherd, D., Burgess, D., Jickells, T., Andrews, J., Cave, R., Turner, R.K., Aldridge, J., Parker, E.R., Young, E., (2007). Modelling the effects and economics of managed realignment on the cycling and storage of nutrients, carbon and sediments in the Blackwater estuary UK. Estuarine, Coastal and Shelf Science, 73(3-4): 355-367.

Shi, C.L., Jin, Z.Q., Ge, D.K., Su, G.L. (2001). Gradual effects of climate change on food production and adaptation strategies in the middle and lower valley of Yangtze River. Jiangsu J of Agr Sci, 17(1):1-6.

Shi, Y.F., Wu, W.L., Meng, F.Q., Wang, D.P., Zhang, Z.H. (2012). Advance in evaluation the effect of carbon sequestration strategies on greenhouse gases mitigation in agriculture. China Population, Resources and Environment, 22(1):43-48.

Sicardi, M., García-Préchac, F., & Frioni, L. (2004). Soil microbial indicators sensitive to land use conversion from pastures to commercial Eucalyptus grandis (Hill ex Maiden) plantations in Uruguay. Applied Soil Ecology, 27(2): 125-133.

Siddique, K.H.M., Regan, K.L., Tennant, D., Thomson, B.D. (2001). Water use and water use efficiency of cool season grain legumes in low rainfall Mediterranean-type environments. European Journal of Agronomy, 15: 267-280.

Siddique, K.H.M., Tennant, D., Perry, M.W., Belford, R.K. (1990). Water use and water use efficiency of old and modern wheat cultivars in a Mediterranean-type environment. Australian Journal of Agricultural Research, 41: 431-477.

Singh, K.B., Malhotra, R.S., Saxena, M.C., & Bejiga, G. (1997). Superiority of winter sowing over traditional spring sowing of chickpea in the Mediterranean region. Agronomy Journal, 89(1): 112-118.

Singh, S.K. & Reddy, K.R. (2011). Regulation of photosynthesis, fluorescence, stomatal conductance and water-use efficiency of cowpea (Vigna unguiculata L. Walp.) under drought. Journal of Photochemistry and Photobiology B-Biology, 105(1): 40-50

Šíp, V., Růžek, P., Chrpová, J., Vavera, R., & Kusá, H. (2009). The effect of tillage practice, input level and environment on the grain yield of winter wheat in the Czech Republic. Field Crops Research, 113(2): 131-137.

Smit, B., & Skinner, M.W. (2002). Adaptation options in agriculture to climate change: a typology. Mitigation and adaptation strategies for global change, 7(1): 85-114.

Smith, J. B., Vogel, J. M., & Cromwell III, J. E. (2009). An architecture for government action on adaptation to climate change. An editorial comment. Climatic Change, 95(1-2), 53-61.

Smith, K.A., Mosier, A.R., Crutzen, P.J., & Winiwarter, W. (2012). The role of N2O derived from crop-based biofuels, and from agriculture in general, in Earth's climate. Philosophical Transactions of the Royal Society B: Biological Sciences, 367(1593): 1169-1174.

Smith, P. (2012). Agricultural greenhouse has mitigation potential globally, in Europe and in the UK: what have we learnt in the last 20 years? Global Change Biology, 18: 35-43.

Smith, P. and T. J. F. Smith (2000). Transport carbon costs do not negate the benefits of agricultural carbon mitigation options. Ecology Letters, 3(5): 379-381.

Smith, P., & Olesen, J.E. (2010). Synergies between the mitigation of, and adaptation to, climate change in agriculture. *Journal of Agricultural Science*, *148*(5): 543-552.

Smith, P., Goulding, K.W., Smith, K.A., Powlson, D.S., Smith, J.U., Falloon, P., & Coleman, K. (2001). Enhancing the carbon sink in European agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. Nutrient Cycling in Agroecosystems, 60(1-3): 237-252.

Smith, P., Goulding, K.W.T., Smith, K.A., Powlson, D.S., Smith, J.U., Falloon, P., & Coleman, K. (2000a). Including trace gas fluxes in estimates of the carbon mitigation potential of UK agricultural land. Soil use and management, 16(4): 251-259.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J. (2008). Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B: Biological Sciences, 363(1492): 789-813.

Smith, P., Milne, R., Powlson, D.S., Smith, J.U., Falloon, P., & Coleman, K. (2000c). Revised estimates of the carbon mitigation potential of UK agricultural land. Soil Use and Management, 16(4): 293-295.

Smith, P., Powlson, D., Glendining, M., & Smith, J.O. (1997). Potential for carbon sequestration in European soils: Preliminary estimates for five scenarios using results from long-term experiments. Global Change Biology, 3(1): 67-79.

Smith, P., Powlson, D.S., Glendining, M.J., & Smith, J.U. (1998). Preliminary estimates of the potential for carbon mitigation in European soils through no-till farming. Global Change Biology, 4(6): 679-685.

Smith, P., Powlson, D.S., Smith, J.U., Falloon, P., & Coleman, K. (2000b). Meeting the UK's climate change commitments: options for carbon mitigation on agricultural land. Soil Use and Management, 16(1): 1-11.

Smithers, R.J., Cowan, C., Harley, M., Hopkins, J.J., Pontier, H., Watts, O., (2008). England Biodiversity Strategy - Climate Change Adaptation Principles: Conserving biodiversity in a changing climate. DEFRA, 16pp.

Smits, A., Nienhuis, P.H., Saeijs, H.L.F., (2006). Changing Estuaries, Changing Views. Hydrobiologia, 565(1): 339-355.

Soane, B.D., Ball, B.C., Arvidsson, J., Basch, G., Moreno, F., & Roger-Estrade, J. (2012). No-till in northern, western and south-western Europe: a review of problems and opportunities for crop production and the environment. Soil and Tillage Research, 118: 66-87.

Sommer, S.G., Olesen, J.E., Petersen, S.O., Weisbjerg, M.R., Valli, L., Rodhe, L., & Béline, F. (2009). Region-specific assessment of greenhouse gas mitigation with different manure management strategies in four agroecological zones. Global Change Biology, 15(12): 2825-2837.

Sommerfeldt, T.G., Chang, C., & Entz, T. (1988). Long-term annual manure applications increase soil organic matter and nitrogen, and decrease carbon to nitrogen ratio. Soil Science Society of America Journal, 52(6): 1668-1672.

Southworth, J., Pfeifer, R. A., Habeck, M., Randolph, J.C., Doering, O.C., & Rao, D.G. (2002). Sensitivity of winter wheat yields in the Midwestern United States to future changes in climate, climate variability, and CO2 fertilization. Climate Research, 22(1): 73-86.

Spala, A., Bagiorgas, H.S., Assimakopoulos, M.N., Kalavrouziotis, J., Matthopoulos, D., & Mihalakakou, G. (2008). On the green roof system. Selection, state of the art and energy potential investigation of a system installed in an office building in Athens, Greece. Renewable Energy, 33(1): 173-177.

Spatari, S., Yu, Z., Montalto, F.A., (2011). Life cycle implications of urban green infrastructure. Environmental Pollution 159(8-9): 2174-2179.

Spencer, K.L., Cundy, A.B., Davies-Hearn, S., Hughes, R., Turner, S., MacLeod, C.L., (2008). Physicochemical changes in sediments at Orplands Farm, Essex, UK following 8 years of managed realignment. Estuarine, Coastal and Shelf Science, 76(3): 608-619.

Speybroeck, J.,Bonte, D., Courtens, W., Gheskiere, T., Grootaert, P., Maelfait, J.-P., Mathys, M., Provoost, S., Sabbe, K., Stienen, E.W.M., Lancker, V.V., Vincx, M., Degraer, S., (2006). Beach nourishment: an ecologically sound coastal defence alternative? A review. Aquatic Conservation: Marine and Freshwater Ecosystems, 16(4): 419-435.

Stalenberg, B. (2012). Innovative flood design in highly urbanised water cities. In: Aerts, J., Botzen, W., Bowman, M.J., Ward, P.J. & Dircke, P. (Eds.) Climate adaptation and flood risk in coastal cities. Abington, Oxon: Earthscan.

Stevens, C.J. and Quinton, J.N. (2009). Diffuse Pollution Swapping in Arable Agricultural Systems. Critical Reviews in Environmental Science and Technology, 39(6): 478-520.

Stewart, A.A., Little, S.M., Ominski, K.H., Wittenberg, K.M., & Janzen, H.H. (2009). Evaluating greenhouse gas mitigation practices in livestock systems: an illustration of a whole-farm approach. The Journal of Agricultural Science, 147(4): 367.

Stockan, J.A., Langan, S.J., Young, M.R. (2012). Investigating riparian margins for vegetation patterns and plant-environment relationships in northeast Scotland. Journal of Environmental Quality, 4(2): 364-372.

Stoorvogel, J.J., Antle, J.M., Crissman, C.C., & Bowen, W. (2004). The tradeoff analysis model: integrated bio-physical and economic modeling of agricultural production systems. Agricultural Systems, 80(1): 43-66.

Stovin, V., Vesuviano, G., & Kasmin, H. (2012). The hydrological performance of a green roof test bed under UK climatic conditions. Journal of Hydrology, 414: 148-161.

Strømann-Andersen, J., & Sattrup, P.A. (2011). The urban canyon and building energy use: Urban density versus daylight and passive solar gains. Energy and Buildings, 43(8): 2011-2020.

Svenning, J.C., Fløjgaard, C., Morueta-Holme, N., Lenoir, J., Normand, S., & Skov, F. (2009). Big moving day for biodiversity? A macroecological assessment of the scope for assisted colonization as a conservation strategy under global warming. In IOP Conference Series: Earth and Environmental Science, 8(1): 012017). IOP Publishing.

Swart, R.J., Biesbroek, G.R., Binnerup, S., Carter, T.R., Henrichs, T., Loquen, S. (2009). Europe Adapts to Climate Change: Comparing National Adaptation Strategies (No. 01). Finnish Environment Institute (SYKE), Helsinki (2009).

SWD (2013). 132 final, Impact assessment – Part 1. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. European Commission, Brussels, 57pp.

SWD (2013). 134 final, Guidelines on developing adaptation strategies. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. An EU strategy on adaptation to climate change. European Commission, Brussels, 54pp.

Sylvester-Bradley, R., Kindred, D.R. (2009). Analysing nitrogen responses of cereals to prioritise routes to the improvements of nitrogen use efficiency. J Exp Bot, 60: 1939–1951.

Symonds, A.M., and Collins, M.B. (2007a). The development of artificially created breaches in an embankment as part of a managed realignment, Freiston Shore, UK. Journal of Coastal Research, 50: 130-134.

Symonds, A.M., and Collins, M.B. (2007b). The establishment and degeneration of a temporary creek system in response to managed coastal realignment: The Wash, UK. Earth Surface Processes and Landforms, 32(12): 1783-1796.

Synnefa, A., Karlessi, T., Gaitani, N., Santamouris, M., Assimakopoulos, D. N., & Papakatsikas, C. (2011). Experimental testing of cool colored thin layer asphalt and estimation of its potential to improve the urban microclimate. Building and Environment, 46(1): 38-44.

Tàbara, J.D. (2011). Integrated Climate Governance (ICG) and Sustainable Development. In European research on sustainable development (pp. 91-109). Springer Berlin Heidelberg.

Tallis, M., Taylor, G., Sinnett, D., & Freer-Smith, P. (2011). Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. Landscape and Urban Planning, 103(2): 129-138.

Tao, F., Yokozawa, M., Zhang, Z., Hayashi, Y., Grassl, H., Fu, C. (2004). Variability in climatology and agricultural production variability in China in association with East Asian monsoon and El Nino Southern Oscillation. Clim. Res. 28: 23-30.

Tao, F.L., Zhang, Z. (2010). Adaptation of maize production to climate change in North China Plain: Quantify the relative contributions of adaptation options. European Journal of Agronomy, 33:103-106.

Teal, J.M. & Weishar, L. (2005). Ecological engineering, adaptive management, and restoration management in Delaware Bay salt marsh restoration. Ecological Engineering, 25(3): 304-314.

Tebrügge, F., & Düring, R.A. (1999). Reducing tillage intensity - a review of results from a long-term study in Germany. Soil and tillage research, 53(1): 15-28.

Teemusk, A., & Mander, Ü. (2010). Temperature regime of planted roofs compared with conventional roofing systems. Ecological Engineering, 36(1): 91-95.

Ten Kate, K.., Bishop, J., and Bayon, R. (2004). Biodiversity offsets: Views, experience, and the business case. IUCN, Gland, Switzerland and Cambridge, UK and Insight Investment, London, UK, 95pp.

Thiere, G., Stadmark, J., Weisner, S.E.B., (2011). Nitrogen retention versus methane emission: Environmental benefits and risks of large-scale wetland creation. Ecological Engineering, 37(1): 6-15.

Thomas, C.D. (2011). Translocation of species, climate change, and the end of trying to recreate past ecological communities. Trends in ecology & evolution, 26(5): 216-221.

Thomas-Hope, E. & Spence, B. (2002). Promoting agrobiodiversity under difficulties: the Jamaica-PLEC experience. PLEC News and Views, 19: 17-24.

Thomson, A.J., Giannopoulos, G., Pretty, J., Baggs, E.M., & Richardson, D.J. (2012). Biological sources and sinks of nitrous oxide and strategies to mitigate emissions. Philosophical Transactions of the Royal Society B: Biological Sciences, 367(1593): 1157-1168.

Tian, Y., Su, D., Li, F., Li, X. (2003). Effect of rainwater harvesting with ridge and furrow on yield of potato in semiarid areas. Field Crops Research, 84: 385-391.

Titto, C.G., Titto, E.A.L., Titto, R.M., & Mourao, G.B. (2011). Heat tolerance and the effects of shade on the behavior of Simmental bulls on pasture. Animal Science Journal, 82(4): 591-600.

Todd, M., Taylor, R., Osborn, T., Kingston, D., Arnell, N. & Gosling, S. (2011). Uncertainty in climate change impacts on basin-scale freshwater resources – preface to the special issue: the QUEST-GSI methodology and synthesis of results. Hydrology and Earth System Sciences, 15: 1035-1046.

Tol, R.S.J. (2005). Adaptation and mitigation: trade-offs in substance and methods. Environmental Science & Policy 8(6): 572-578.

Tomer, M.D. & Schilling, K.E. (2009). A simple approach to distinguish land-use and climate-change effects on watershed hydrology. Journal of Hydrology, 376: 24-33.

Tompkins, E.L., Adger, W.N., Boyd, E., Nicholson-Cole, S., Weatherhead, K. & Arnell, N. (2010). Observed adaptation to climate change: UK evidence of transition to a well-adapting society. Global Environmental Change, 20: 627–635.

Tompkins, E.L., Few, R., & Brown, K. (2008). Scenario-based stakeholder engagement: incorporating stakeholders preferences into coastal planning for climate change. Journal of Environmental Management, 88(4): 1580-1592.

Tonietto, R., Fant, J., Ascher, J., Ellis, K., & Larkin, D. (2011). A comparison of bee communities of Chicago green roofs, parks and prairies. Landscape and Urban Planning, 103(1): 102-108.

Tovey, E.L., Pontee, N. I. Harvey, R. (2009). Managed realignment at Hesketh out Marsh West. Proceedings of the Institution of Civil Engineers. Engineering sustainability, 162(4): 223-228.

Townend, I. and J. Pethick (2002). Estuarine Flooding and Managed Retreat. Philosophical Transactions: Mathematical, Physical and Engineering Sciences, 360(1796): 1477-1495.

Trabucco, A., Zomer, R.J., Bossio, D.A., van Straaten, O. & Verchot, L.V. (2008). Climate change mitigation through afforestation/reforestation: A global analysis of hydrologic impacts with four case studies. Agriculture, Ecosystems & Environment, 126: 81-97.

Triberti, L., Nastri, A., Giordani, G., Comellini, F., Baldoni, G., & Toderi, G. (2008). Can mineral and organic fertilization help sequestrate carbon dioxide in cropland? European Journal of Agronomy, 29(1): 13-20.

Trnka, M., Dubrovský, M., & Žalud, Z. (2004). Climate change impacts and adaptation strategies in spring barley production in the Czech Republic. Climatic Change, 64(1-2): 227-255.

Trnka, M., Eitzinger, J., Dubrovský, M., Semerádová, D., Štěpánek, P., Hlavinka, P., Balek, J., Skalák, Farda, A., Formayer, H., Žalud, Z. (2010). Is rainfed crop production in central Europe at risk? Using a regional climate model to produce high resolution agroclimatic information for decision makers. The Journal of Agricultural Science, 148(6): 639-656.

Trulio, L., Callaway, J., Crooks, S., (2007). White Paper on Carbon Sequestration and Tidal Salt Marsh Restoration, 4pp.

Tsihrintzis, V.A., Vasarhelyi, G.M., Lipa, J. (1996). Ballona Wetland: A multi-objective salt marsh restoration plan. Proceedings of the ICE - Water Maritime and Energy, 118(2): 131-144

Tubiello, F.N., Donatelli, M., Rosenzweig, C., & Stockle, C.O. (2000). Effects of climate change and elevated CO2 on cropping systems: model predictions at two Italian locations. European Journal of Agronomy, 13(2): 179-189.

Tubiello, F.N., Rosenzweig, C., Goldberg, R.A., Jagtap, S., & Jones, J.W. (2002). Effects of climate change on US crop production: simulation results using two different GCM scenarios. Part I: Wheat, potato, maize, and citrus. Climate Research, 20(3): 259-270.

Tuong, T.P., Bouman, B.A.M., Mortimer, M. (2005). More rice, less water-integrated approaches for increasing water productivity in irrigated rice-based systems in Asia. Plant Prod Sci, 8:231-241.

Turner, N.C. (2004a). Sustainable production of crops and pastures under drought in a Mediterranean environment. Annals of Applied Biology, 144: 139-147.

Turner, N.C. (2004b) Agronomic options for improving rainfall use efficiency of crops in dryland farming systems. Journal of Experimental Botany, 55: 2413-2425.

Turner, N.C., Asseng, S. (2005). Productivity, sustainability and rainfall-use efficiency in Australian rainfed Mediterranean agricultural systems. Australian Journal of Agricultural Research, 56: 1123-1136.

Turner, N.C., Molyneux, N., Yang, S., Xiong, Y.C., & Siddique, K.H. (2011). Climate change in south-west Australia and north-west China: challenges and opportunities for crop production. Crop and Pasture Science, 62(6): 445-456.

Turner, N.C., Wright, G.C., Siddique, K.H.M. (2001). Adaptation of grain legumes (pulses) to water-limited environments. Advances in Agronomy, 71: 193-231.

UKCIP (2010a). Linking biodiversity action plan and flood alleviation: Selby case study. UK Climate Impacts Programme, 2pp. Available: http://www.ukcip.org.uk/wordpress/wp-content/LA_casestudies/LA_CS_Selby.pdf Accessed: July 2013.

UKCIP (2010b). West Midlands case study. UK Climate Impacts Programme, 3pp. Available: http://www.ukcip.org.uk/wordpress/wp-content/LA_casestudies/LA_CS_WestMidlands.pdf Accessed: July 2013.

UKCIP (2010c). Improving connectivity of habitats to increase resilience for biodiversity: Kent. UK Climate Impacts Programme, 3pp. Available: http://www.ukcip.org.uk/wordpress/wp-content/LA_casestudies/LA_CS_Kent_biodiversity.pdf Accessed: July 2013.

UNFCCC (2008a). Database on Local Coping Strategies: Growing acacia albida in Burkina Faso. Available at: http://maindb.unfccc.int/public/adaptation/adaptation_casestudy.pl?id_project=100 Accessed: July 2012.

UNFCCC (2008b). Database on Local Coping Strategies: Reforestation/Afforestation to prevent soil erosion and land slides in Tajikistan. Available at: http://maindb.unfccc.int/public/adaptation/adaptation_casestudy.pl?id_project=179 Accessed: July 2012.

UNFCCC (2008c). Database on Local Coping Strategies: Mangrove reforestation in southern Thailand. Available at: http://maindb.unfccc.int/public/adaptation/adaptation_casestudy.pl?id_project=154 Accessed: July 2012.

Urwin, K., Jordan, A. (2008). Does public policy support or undermine climate change adaptation? Exploring policy interplay across different scales of governance. Global Environmental Change, 18: 180-191.

Vadez, V., Kholova, J., Choudhary, S., Zindy, P., Terrier, M., Krishnamurthy, L., Kumar, P.R., Turner, N.C. (2011). Responses to increased moisture stress and extremes: whole plant response to drought under climate change. Crop adaptation to climate change: 186-196.

Van Bodegom A.J., Avenije H., Wit M. (2009). Forests and Climate Change: adaptation and mitigation. Tropenbos International, Wageningen, the Netherlands: xvi + 160 pp.

Van den Putte, A., Govers, G., Diels, J., Gillijns, K., & Demuzere, M. (2010). Assessing the effect of soil tillage on crop growth: A meta-regression analysis on European crop yields under conservation agriculture. European Journal of Agronomy, 33(3): 231-241.

Van der Sluis, T., Bloemmen, M., & Bouwma, I.M. (2004). European corridors: Strategies for corridor development for target species. ECNC & Alterra (Netherlands), 34pp.

Van Kessel, C. and C. Hartley (2000). Agricultural management of grain legumes: has it led to an increase in nitrogen fixation? Field Crops Research, 65(2–3): 165-181.

Van Minnen, J. G., Strengers, B. J., Eickhout, B., Swart, R., & Leemans, R. (2008). Quantifying the effectiveness of climate change mitigation through forest plantations and carbon sequestration with an integrated land-use model. Carbon Balance and Management, 3(3): 20.

Van Proosdij, D., Lundholm, J., Neatt, N., Bowron, T., Graham, J., (2010). Ecological re-engineering of a freshwater impoundment for salt marsh restoration in a hypertidal system. Ecological Engineering, 36(10): 1314-1332.

Van Roon, M.R. (2012). Wetlands in The Netherlands and New Zealand: Optimising biodiversity and carbon sequestration during urbanisation. Journal of Environmental Management, 101: 143-150.

Van Vuuren, D.P., Isaac, M., Kundzewicz, Z.W., Arnell, N., Barker, T., Criqui, P., Berkhout, F., Hilderink, H., Hinkel, J., Hof, A., Kitous, A., Kram, T., Mechler, R. & Scrieciu, S. (2011). The use of scenarios as the basis for combined assessment of climate change mitigation and adaptation. Global Environmental Change, 21(2): 575-591.

Vandenbroeck, J. (2006). Port 2000, Le Havre's new container terminal: breakwaters and dredging of the nautical access channel. Terra et Aqua 104: 22.

Verbessem, I., van den Bergh, E., de Regge, N., Soors, J., de Belder, W., de Groot, R., (2007). Sediment characteristics and sedimentation-erosion processes on Ketenisse polder one year after levelling. Aquatic Ecosystem Health & Management, 10(1): 87-92.

Verch, G., Kächele, H., Höltl, K., Richter, C., & Fuchs, C. (2009). Comparing the profitability of tillage methods in Northeast Germany - A field trial from 2002 to 2005. Soil and Tillage Research, 104(1): 16-21.

Viguié, V., & Hallegatte, S. (2012). Trade-offs and synergies in urban climate policies. Nature Climate Change, 2(5): 334-337.

VijayaVenkataRaman, S., Iniyan, S., Goic, R. (2012). A review of climate change, mitigation and adaptation. Renewable and Sustainable Energy Reviews, 16(1): 878-897.

Villarreal, E.L., Semadeni-Davies, A., & Bengtsson, L. (2004). Inner city stormwater control using a combination of best management practices. Ecological Engineering, 22(4): 279-298.

Vitt, P., Havens, K., Kramer, A.T., Sollenberger, D., & Yates, E. (2010). Assisted migration of plants: changes in latitudes, changes in attitudes. Biological Conservation, 143(1): 18-27.

Vleeshouwers, L.M., Verhagen, A. (2002). Carbon emission and sequestration by agricultural land use: a model study for Europe. Global Change Biology, 8(6): 519-530.

Voldseth, R.A., Johnson, W.C., Guntenspergen, G.R., Gilmanov, T., & Millett, B.V. (2009). Adaptation of farming practices could buffer effects of climate change on northern prairie wetlands. Wetlands, 29(2): 635-647.

Von Storch, H., Gönnert, G., Meine, M., (2008). Storm surges--An option for Hamburg, Germany, to mitigate expected future aggravation of risk. Environmental Science & Policy, 11(8): 735-742.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Reidy Liermann, C. & Davies, P.M. (2010). Global threats to human water security and river biodiversity. Nature, 467: 555-561.

Vos, C.C., Berry, P., Opdam, P., Baveco, H., Nijhof, B., O'Hanley, J., Bell, C. & Kuipers, H. (2008). Adapting landscapes to climate change: examples of climate-proof ecosystem networks and priority adaptation zones. Journal of Applied Ecology, 45(6): 1722-1731.

Wagner, R. G., Little, K. M., Richardson, B., & Mcnabb, K. (2006). The role of vegetation management for enhancing productivity of the world's forests.Forestry, 79(1), 57-79.

Walker, A.K., & Campbell, J. (2010). Marine fungal diversity: a comparison of natural and created salt marshes of the north-central Gulf of Mexico. Mycologia, 102(3): 513-521.

Walker, S., Brower, A.L., Stephens, R.T., & Lee, W.G. (2009). Why bartering biodiversity fails. *Conservation Letters*, *2*(4): 149-157.

Walley, F.L., Clayton, G.W., Miller, P.R., Carr, P.M., & Lafond, G.P. (2007). Nitrogen economy of pulse crop production in the Northern Great Plains. Agronomy Journal, 99(6): 1710-1718.

Walsh, C.L., Dawson, R.J., Hall, J.W., Barr, S.L., Batty, M., Bristow, A.L., Carney, S., Dagoumas, A.S., Ford, A.C., Harpham, C., Tight, M.R., Watters, H., Zanni, A.M. (2010). Assessment of climate change mitigation and adaptation in cities. Proceedings of the ICE-Urban Design and Planning, 164(2): 75-84.

Wang, C.Y., Zhao, J.B., Li, X.Y. (2006). Study on agricultural adaptation to warming and drying climate in North China. Arid Land Geography, 29(5):646-652.

Wang, J., Huang, J., Rozelle, S. (2010). Climate change and China's agricultural sector: an overview of impacts, adaptation and mitigation. Adaptation and Mitigation, ICTSD–IPC Platform on Climate Change, Agriculture and Trade, Issue Brief 5, International Centre for Trade and Sustainable Development, Geneva, Switzerland and International Food & Agricultural Trade Policy Council, Washington DC, USA.

Wang, X.B., Wu, X.P., Zhao, Q.S., Deng, X.Z., Cai, D.X., (2011). Effects of cropland-use management on potentials of soil carbon sequestration and carbon emission mitigation in China. Scientia Agricultura Sinica, 44(11): 2284-2293.

Wang, X.F., Chen, B.M., Bi, J.Y. (2012). Assessment of Agricultural water resources' production efficiency at county level in China. Journal of Irrigation and Drainage, 31(3): 6-10.

Wang, Y.C., Ma, S.M. (2009). Technological options of regional agricultural adaptation to climate change in China. Chinese Journal of Agrometeorology, 30(1):51-56. (In Chinese with English abstract)

Warren, R.S., Fell, P.E., Rozsa, R., Brawley, A.H., Orsted, A.C., Olson, E.T., Swamy, C., Niering, W.A., (2002). Salt marsh restoration in Connecticut: 20 years of science and management. Restoration Ecology, 10(3): 497-513.

Weckwerth, W. (2011). Green systems biology - From single genomes, proteomes and metabolomes to ecosystems research and biotechnology. Journal of Proteomics 75(1): 284-305.

Weckwerth, W. (2011). Green systems biology—from single genomes, proteomes and metabolomes to ecosystems research and biotechnology. Journal of proteomics, 75(1): 284-305.

Weiske, A., Vabitsch, A., Olesen, J.E., Schelde, K., Michel, J., Friedrich, R., & Kaltschmitt, M. (2006). Mitigation of greenhouse gas emissions in European conventional and organic dairy farming. Agriculture, ecosystems & environment, 112(2): 221-232.

Wells, M.J. and T. Turpin (1999). Creating a Wetland Using Rainwater as Mitigation for Loss of Waterfowl Habitat. Water and Environment Journal, 13(3): 157-163.

Wen, X.X., Weng, Q., Li, S.Y., Yang, G.H. (2003). Development and utilization of alfalfa prataculture on the Loess Plateau. Agricultural Research in the Arid Areas, 21: 160-164. (In Chinese with English abstract).

Wende, W., Bond, A., Bobylev, N., & Stratmann, L. (2012). Climate change mitigation and adaptation in strategic environmental assessment. Environmental Impact Assessment Review, 32(1): 88-93.

West, C., Gawith, M. (2005). Measuring progress: Preparing for climate change through the UK Climate Impacts Programme. UKCIP technical report, Oxford, 72pp.

West, J.W. (2003). Effects of heat-stress on production in dairy cattle. Journal of Dairy Science, 86(6): 2131-2144.

West, T. O., & Marland, G. (2003). Net carbon flux from agriculture: Carbon emissions, carbon sequestration, crop yield, and land-use change. Biogeochemistry, 63(1): 73-83.

West, T.O. & Post, W.M. (2002). Soil organic carbon sequestration rates by tillage and crop rotation. Soil Sci. Soc. Am. J. 66(6): 1930-1946.

Whalley, W.R., Clark, L.J., Gowing, D.J.G., Cope, R.E., Lodge, R.J., & Leeds-Harrison, P. B. (2006). Does soil strength play a role in wheat yield losses caused by soil drying? *Plant and Soil*, *280*(1-2): 279-290.

Wilbanks, T J., Sathaye, J. (2007). Integrating mitigation and adaptation as responses to climate change: a synthesis. Mitigation and Adaptation Strategies for Global Change, 12(5): 957-962.

Wilbanks, T.J., & Kates, R.W. (1999). Global change in local places: how scale matters. Climatic Change, 43(3): 601-628.

Wilbanks, T.J., Leiby, P., Perlack, R., Ensminger, J.T., & Wright, S.B. (2007). Toward an integrated analysis of mitigation and adaptation: some preliminary findings. Mitigation and Adaptation Strategies for Global Change, 12(5): 713-725.

Wilby, R.L., & Dessai, S. (2010). Robust adaptation to climate change. Weather, 65(7): 180-185.

Williams, K. (1999). Urban intensification policies in England: problems and contradictions. Land Use Policy, 16(3): 167-178.

Willis, S.G., Hill, J.K., Thomas, C.D., Roy, D.B., Fox, R., Blakeley, D.S., & Huntley, B. (2009). Assisted colonization in a changing climate: a test-study using two UK butterflies. Conservation Letters, 2(1): 46-52.

Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A. & Morris, M. (2011). The impact of drain blocking on an upland blanket bog during storm and drought events, and the importance of sampling-scale. Journal of Hydrology, 404: 198-208.

Winn, P.J.S., Young, R.M., Edwards, A.M.C., (2003). Planning for the rising tides: the Humber Estuary Shoreline Management Plan. The Science of the Total Environment 314: 13-30.

Wise S., Ghalayini, D.J.B., Grant, J., Kloss, C., MacMullan, E., Morse, F.M., Nees, D. Nowak, D., Peck, S. Shaikh, S., Yu, C. (2010). Integrating valuation methods to recognize green infrastructure's multiple benefits. Low Impact Development: 1123-1143.

Witcombe, J.R., Hollington, P.A., Howarth, C.J., Reader, S., & Steele, K.A. (2008). Breeding for abiotic stresses for sustainable agriculture. Philosophical Transactions of the Royal Society B: Biological Sciences, 363(1492): 703-716.

Wojdyga, K. (2009). An investigation into the heat consumption in a low-energy building. Renewable energy, 34(12): 2935-2939.

Wolfe, M.S., Baresel, J.P., Desclaux, D., Goldringer, I., Hoad, S., Kovacs, G., Löschenberger, F., Miedaner, T., Østergård, H., Lammerts Van Bueren, E.T. (2008). Developments in breeding cereals for organic agriculture. Euphytica, 163(3): 323-346.

Wolff, W.J. (1992). The end of a tradition: 1000 years of embankment and reclamation of wetlands in the Netherlands. Ambio, 21(4): 287-291.

Wolters, M., Garbutt, A., Bakker, J.P., (2005a). Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. Biological Conservation, 123(2): 249-268.

Wolters, M., Garbutt, A., Bakker, J.P., (2005b). Plant colonization after managed realignment: the relative importance of diaspore dispersal. Journal of Applied Ecology, 42(4): 770-777.

Wolters, M., Garbutt, A., Bekker, R.M., Bakker, J.P., Carey, P.D., (2008). Restoration of salt-marsh vegetation in relation to site suitability, species pool and dispersal traits. Journal of Applied Ecology, 45(3): 904-912.

Woodward, R.T., & Wui, Y.-S. (2001). The economic value of wetland services: a meta-analysis. Ecological Economics, 37(2): 257-270.

World Agroforestry Centre (2007). Advancing the science and practice of agroforestry to help realise an agroforestry transformation throughout the developing world. Nairobi, Kenya.

WRD (2011). De Doorbraak Project. Available: http://www.wrd.nl/dedoorbraak/algemene_onderdelen/english Accessed: June, 2013.

Wu, J.J., He, B., Lu, A.F., Zhou, L., Liu, M., Zhao, L. (2011). Quantitative assessment and spatial characteristics analysis of agricultural drought vulnerability in China. Nat Hazards, 56: 785-801.

WWF (2002). Abotts Hall Farm: coastal wetland restoration on a working farm. Marine Update 52, 6pp.

Xie, L.Y., Li, Y., Lin, M. (2011). Response and adaptation to climate change of agriculture and environment in Northeast China. Chinese Journal of Eco-Agriculture, 19(1): 197-201. (in Chinese with English abstract)

Xing, Y., Hewitt, N., Griffiths, P., (2011). Zero carbon buildings refurbishment - a Hierarchical pathway. Renewable and Sustainable Energy Reviews, 15(6): 3229-3236.

Xiong, H.Y., Li, T.X., Zhang, X.Z., Yu, H.Y. (2008). Amount changes of microorganism and microbial biomass of no-tillage paddy soil after paddy-upland rotation. Soils, 40(6): 920-925.

Xiong, W., Holman, I., Lin, E., Conway, D., Jiang, J., Xu, Y., Li, Y. (2010). Climate change, water availability and future cereal production in China. Agriculture, Ecosystems and Environment, 135: 58-69.

Xu, J.Z., Peng, S.Z., Hou, H.J., Yang, S.H., Luo, Y.F., Wang, W.G. (2012). Gaseous losses of nitrogen by ammonia volatilization and nitrous oxide emissions from rice paddies with different irrigation management. Irrig Sci: 1-12.

Yan, H.M., Cao, M.K., Liu, J.Y., Tao, B. (2007). Potential and sustainability for carbon sequestration with improved soil management in agricultural soils of China. Agriculture, Ecosystems and Environment, 121: 325-335.

Yang, J., Shen, Y.Y., Nan, Z.B., Gao, C.Y., Niu, Y.N.,Wang, X.Z., Luo, C.Y., Li, G.D. (2010). Effects of conservation tillage on crop yield and carbon pool management index on top soil within a maize-wheat-soy rotation system in the Loess Plateau. Acta Prataculturae Sinica, 19: 75-82. (In Chinese with English abstract).

Yang, Y.M., Yang, Y.G., Moiwo, J.P., Hu, Y. (2010). Estimation of irrigation requirement for sustainable water resources reallocation in North China. Agricultural Water Management, 97(11): 1711-1721

Yau, S.K. (2007). Winter versus spring sowing of rain-fed safflower in a semi-arid, high-elevation Mediterranean environment. European Journal of Agronomy, 26(3): 249-256.
York, pp. 145–163

You, S.C. (2001). Agricultural adaptation to climate change in China. J Environ Sci China 13: 192-197.

Yu, O.T. & Chmura, G.L. (2009). Soil carbon may be maintained under grazing in a St Lawrence Estuary tidal marsh. Environmental Conservation, 36(4): 312-320.

Yuan, J., Xu, Y.L. (2008). Study on adaptation measures of wheat production in Linyi of Shandong province based on CERES crop models. Chinese Journal of Agrometeorology, 29(3): 251-255 (in Chinese with English abstract).

Zhang, B., Zhang, N., Zhang, T.F. (2011). Performance evaluation of adaptation measures to climate change in agriculture. Agricultural Technical Economy, 7: 43-49.

Zhang, J.P., Zhao, Y.X., Wang, C.Y., He, Y. (2006). Effects of climate change on winter wheat growth and yield in North China. Chinese Journal of Applied Ecology, 17(7): 1179-1181.

Zhang, Y.S., Cai, R.S., Fu, L.L., Liu, L.J., Dong, H.F. (2012). Greenhouse gas emissions from major agricultural activities in China and corresponding mitigation strategies. Journal of Zhejiang University (Agric & Life Sci.), 38(1): 97-107.

Zhou, L.M., Li, F.M., Jin, S.L., Song, Y. (2009). How two ridges and the furrow mulched with plastic film affect soil water, soil temperature and yield of maize on the semiarid Loess Plateau of China. Field Crops Research, 113: 41-47.

Zhou, S.D., Zhou, W.K., Zhu, H.G., Wang, C.X., Wang, Y. (2010). Impact of climate change on agriculture and its countermeasures. Journal of Nanjing Agricultural University (Social Sciences Edition), 10(1): 34-39 (in Chinese with English abstract).

Zimmerman, M., & Andersson, J. (1998). Low energy cooling: Case study buildings. IEA: Energy conservation in buildings and community systems programme, Annex 28 - Low Energy Cooling, 159pp.

Zinzi, M., & Agnoli, S. (2011). Cool and green roofs. An energy and comfort comparison between passive cooling and mitigation urban heat island techniques for residential buildings in the Mediterranean region. Energy and Buildings, 55: 66-76.

Zmelik, K., Schindler, S., & Wrbka, T. (2011). The European Green Belt: international collaboration in biodiversity research and nature conservation along the former Iron Curtain. Innovation: The European Journal of Social Science Research, 24(3): 273-294.

Zou, J.W., Huang, Y., Zheng, X.H., Wang, Y. (2007). Quantifying direct N_2O emissions in paddy field during rice growing season in mainland China: Dependence on water regime. Atmosphere Environment, 41(37): 8030-8042.

Zou, X.X., Li, Y.E., Gao, Q.Z., Wan, Y.F. (2012). How water saving irrigation contributes to climate change resilience—a case study of practices in China. Mitig Adapt Strateg Glob Change, 17: 111-132.

Zou, X.X., Li, Y.E., Gao, Q.Z., Wan, Y.F., Shi, S.W. (2011). How to reduce greenhouse gas (GHG) emissions in agriculture: An analysis of measures and actions taken in China. Ecology and Environmental Sciences, 20(8-9): 1348-1358.