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WATER-LEVEL FLUCTUATIONS

Climate change and lakeshore conservation: a model and review of management techniques

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Abstract Climate change is expected to cause significant changes to the hydrology of lakes, reservoirs and other wetlands. In particular, there will be an increase in the level of disturbance produced by water-level fluctuations. This may have adverse consequences for biodiversity, water quality and human uses. Strategies to cope with these climate change impacts are currently poorly developed. This article proposes the use of Grime's CSR theory as a framework to understand the potential impacts of climate change on shoreline vegetation. It is also used to recommend a series of practical management techniques that will contribute to the adaptation capacity of shoreline ecosystems. Four key areas are highlighted: hydrological controls, substrate conditions, shoreline topography and vegetation establishment.

Keywords Shoreline conservation · CSR model · Climate change · Adaptation strategies · Aquatic macrophytes · Water-level fluctuations

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Ecological Effects of Water-Level Fluctations in Lakes

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Introduction

The influence of hydrology on the shoreline vegetation of lakes and reservoirs has long been recognized (e.g. Preston, 1895; Pearsall, 1920). Despite this, many authors have identified an urgent requirement for continued research into the relationship between water-level fluctuations and littoral vegetation dynamics (Levine & Willard, 1990; Zedler & Weller, 1990; Merritt, 1994; Middleton, 1995). Equally important, however, is the need for such research to be synthesized and developed into specific applied guidance for the practitioners whose role it is to conserve lakes, reservoirs and other wetlands (Nilsson & Keddy, 1988; Kusler & Kentula, 1990; Keddy, 1999; Halse & Massenbauer, 2005). A key tool for this knowledge transfer is the development of simple and clear conceptual models for nature conservation managers (Steel & Duncan, 1999; Ogden et al., 2005). Based upon ecological theory and empirical evidence, they can help to define management issues, identify trends and processes and allow the development of strategies for practical implementation (Keddy & Fraser, 2000). Examples of such models in the field of wetland ecology are centrifugal organization (Wisheu & Keddy, 1992), the flood pulse concept (Middleton, 2002), the wetland continuum (Euliss et al., 2004) and the succession model proposed by van der Valk (1981).

This article proposes an application of Grime's (1979) CSR theory to model the potential impacts of climate change on shoreline vegetation. This conceptual framework is then used to provide a context for a series of practical strategies that can be implemented to mitigate the adverse impacts of climate change, i.e. adaptation *sensu* IPCC (2001)—"adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities".

Climate change and water-level fluctuations

Although there is still considerable uncertainty about the potential impacts of climate change on freshwater ecosystems, understanding has developed significantly in recent years. Impacts will include increased water temperatures, sedimentation and pollution of wetland systems and heightened nutrient levels (Hossell et al., 2000). There will also be major impacts on hydrology. Annual average streamflows are expected to increase in northern Europe and decrease in southern and central Europe, but in areas where snowfall has normally been a large component of precipitation, increasing temperatures will result in more winter runoff from rain and less snow-melt during spring. In Britain, northwest regions will have an increased positive water balance, with wetlands having to adapt to higher water levels and increased flooding, while in the southeast a net decrease will cause more frequent and severe summer droughts (Dawson et al., 2001). Due to the general increase in seasonal variability in precipitation, river flows and evapotranspiration, the influence of climate change is likely to have significant impacts on the magnitude, timing and variability of the hydroperiod of waterbodies and the frequency and nature of extreme flood or drought events (Lins et al., 1991; Arnell, 1999; Cannell et al., 1999; Schindler, 2001; Kundzewicz et al., 2002; Fowler et al., 2003; Eisenreich, 2005). Although this may create new habitat in the form of temporary pools or riverine wetlands in areas with increased winter precipitation, the effect for the majority of existing wetland systems will be an increase in the incidence and scale of summer drawdown (Dawson et al., 2001). Currently stable waterbodies will develop into two-phase environments with increasingly separate submersed and terrestrial conditions. This will increase the area of bare substrate that is seasonally exposed around waterbodies and made available as potential habitat for marginal species (Salisbury, 1970). However, the ability of plants to colonize and become established in this newly available habitat will depend upon the frequency, timing and duration of drawdown events. Conversely, the existing habitat of submerged and obligate aquatic species, with stable water levels, will become increasingly unsuitable to support these types of vegetation. Given these potential interactions, changes to water level regimes are likely to have a more critical impact upon the structure, function and biodiversity interest of wetland communities than temperature increases *per se* (van Dam et al., 2002; Hulme, 2005). However, despite this potential for considerable change and the major importance of wetlands as habitat for a diverse suite of rare plant species, little attention has been given, thus far, to the impacts of climate change on wetland vegetation by scientists and policy makers (Dawson et al., 2001). The studies that have been carried out also focus primarily on rivers or groundwater wetlands, to the exclusion of open water bodies such as lakes and reservoirs. One exception to this has been the study of prairie potholes in the USA (e.g. Johnson et al., 2004), which indicates that water levels would be much lower under climate change scenarios, with some wetlands being without standing water for considerable periods of time. Major changes to vegetation would be likely to result, with consequent impacts for other wildlife such as waterfowl.

The CSR model

The classification of species according to their functional characteristics provides a useful context to investigate relationships between vegetation and environmental parameters. Functional criteria, such as leaf and shoot morphology, seed production and growth rates, can be used either as individual attributes or in groups, to define assemblages of plants and enable predictions to be made of changes in vegetation type and species composition (Willby et al., 2000). This approach can, therefore, allow the impacts of events or trends like global climate change to be predicted and assessed.

The key mechanism linking increased water-level fluctuations from climate change with impacts on shoreline vegetation communities is likely to be the disturbance regime (*sensu* Grime, 1979) produced by repeated drawdown and re-flooding on lakeshores. When combined with a fertility gradient (e.g. Day et al., 1988), this can be understood in terms of Grime's CSR theory, which proposes a three-way classification of plant life histories, dividing species into competitive, stress-tolerant or ruderal groups depending on their observed traits. These different classes are arranged on the two opposing environmental axes, one describing the level of habitat disturbance, the other its fertility. These two axes produce four possible types of environment, three of which are inhabitable by plants. The characteristic vegetation of each environmental type has a corresponding life-history: (i) low disturbance, fertile habitats favour competitive species, (ii) low disturbance, low fertility habitats favour stress-tolerant species and (iii) high disturbance, fertile habitats support ruderal species (Fig. 1). The fourth type of environment, with high disturbance and low fertility, is uninhabitable (Grime, 1979).

Shorelines are particularly appropriate subjects for the application of CSR theory, as the two key axes are often clearly expressed in the ecology of their vegetation: variation in environmental parameters causing stress or disturbance is often coarse-grained in scale, and subsequent differences among plant communities are usually well-defined (Spence, 1982; Keddy, 1983). In addition, changes in environment over a relatively short period of time can cause significant differences among communities, for example, as seen in a shift from a competitive marginal community to a ruderal annual vegetation with drawdown. However, these broad changes may be complicated by issues of phenotypic plasticity and the wide ecological amplitude of some wetland species, which may make clear predictions difficult (Willby et al., 2000).

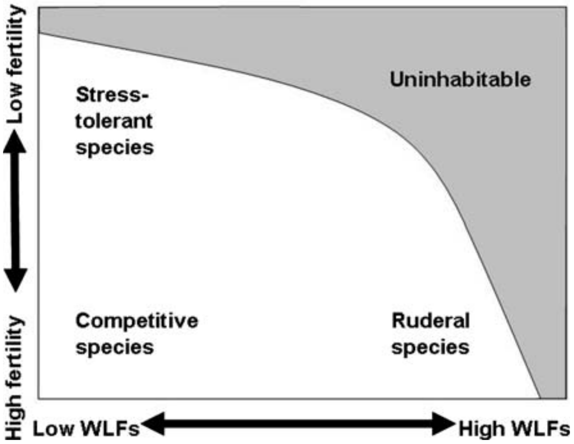

Fig. 1 Grime's CSR triangle applied to shoreline situations, with fertility and disturbance axes shown (WLFs—water-level fluctuations)

There are a number of potential sources of disturbance on lake shorelines, including wave action (Grelsson & Nilsson, 1991), ice scour (Begin & Payette, 1991) and herbivory (Hoffman & Stanley, 1978). However, a number of studies have identified water-level fluctuations as the key component of disturbance in terms of its influence on littoral vegetation dynamics (Gill & Bradshaw, 1971; Nilsson, 1981; Gasith & Gafny, 1990; Irwin & Noble, 1996; Fraisse et al., 1997; Hroudova & Zakravsky, 1999; Keddy, 2000). Drawdown can cause mortality of aquatic plants through heating and desiccation in summer, or freezing in winter (Blindow, 1992; Irwin & Noble 1996) and can allow colonization by a new suite of plant species (Salisbury, 1970). Flooding of terrestrial plants can also reduce cover by preventing respiration, reducing light levels required for photosynthesis, or by initiating chemical changes in the substrate with which they cannot cope (Middleton, 1995). The effects of these changes are consistent with Class I type disturbance (*sensu* Sousa, 1984), increasing resource availability, causing the removal of dominant species and setting back successional processes. As discussed above, climate change will alter hydrological patterns, with increased water-level fluctuations producing a heightened disturbance regime (Mooij et al., 2005).

On the secondary axis of habitat fertility, the level of environmental stress on shorelines is principally dictated by the underlying geology. In addition, exposure to wave action causes erosion, transport, sorting and deposition of sediments, having a major impact on the levels of organic matter, nutrients and fine particles in the substrate (Nilsson, 1981; Keddy, 1984; Wilson & Keddy, 1988; Fraisse et al., 1997). Hence, on wave-washed shores the substrate is often sandy or stony with low nutrient levels, while more sheltered areas have an organic, nutrient-rich soil with a higher proportion of silt and clay (Wilson & Keddy, 1986; Grelsson & Nilsson, 1991). The change in species complement and communities along a fertility gradient is well established (Wilson & Keddy, 1986, 1988) and has impacts upon biomass, species richness and the presence of scarce plants.

The occurrence of these two gradients is illustrated in the study by Willby et al. (2000), which analysed the habitat utilization of 120 hydrophyte species. It found that there were two key axes in habitat parameters relating to (a) flow, substratum grade, organic matter content, scour and sedimentation (a gradient from sheltered bays with fine, mixed or organic sediments to exposed lake shores with coarse-grained mineral strata), and (b) depth, water level stability and biotic disturbance (a gradient from stable, rarely disturbed sites to fluctuating sites). In addition, Hawes et al. (2003) found that the extent and diversity of shallow-growing species was related to a combination of the extent of water-level fluctuation and wave exposure. In their study of riverine marshes, Day et al. (1988) distinguished between five life-history types that broadly parallel the CSR system: clonal dominants, gap colonizers, stress tolerators, reeds and ruderals; which all occur at different positions along gradients of fertility and disturbance.

Fertile, sheltered shores, which allow the retention of organic matter, silt and clay and are not subject to frequent fluctuations (with vegetation gaps forming infrequently), promote the dominance of competitive species. This is ‘core’ habitat according to the centrifugal organization hypothesis, which is occupied by a restricted number of large leafy species capable of forming dense canopies (Wisheu & Keddy, 1992). Species characteristics that accompany this competitive dominance include high biomass, long generation times, low reproductive output and the capacity for vegetative spread. In these habitats, weaker competitors are excluded, with low species richness being a usual result (Day et al., 1988; Shipley et al., 1989; Keddy & Fraser, 2000; Willby et al., 2000). Examples of typical competitive species are lesser bulrush *Typha angustifolia* L., broadfruit bur-reed *Sparganium eurycarpum* Engelm. ex Gray and sweet flag *Acorus calamus* L. Outside of the core habitat in which these species occur, different gradients of stress and disturbance radiate, with varied groups of species and vegetation arranged peripherally along each.

On infertile, wave-eroded shorelines, with low substrate organic matter and high percentages of sand and gravel, only distinctive stress-tolerant species can be supported (Keddy & Fraser, 2000). Small, slow-growing rosette species like isoetids, with anchored tubular leaves that may be evergreen or wintergreen, are typical of this type of habitat (Wilson & Keddy, 1986; Willby et al., 2000). Typical species are often short, low in biomass, have little capacity for vegetative spread, low growth rates and large seeds with slow germination. Examples are American bulrush *Schoenoplectus americanus* (Pers.) Volk. ex Schinz & Keller, common spike-rush *Eleocharis palustris* (L.) Roem. & Schult., bald spike-rush *Eleocharis erythropoda* Steud. and bugleweed *Lycopus uniflorus* Michx (Shipley et al.,1989). Moore et al. (1989) also found that these infertile shores support a particularly high proportion of rare species, and therefore, have high nature conservation value.

On fertile shorelines subject to disturbance from frequent water-level fluctuations, with frequent and density-independent mortality, ruderal plants such as celery-leaved buttercup *Ranunculus sceleratus* L., trifid bur-marigold *Bidens tripartita* L. and marsh cudweed *Gnaphalium uliginosum* L. would be expected (Ellenberg, 1988). Fast growing annuals or short-lived perennials are often typical of this habitat, displaying characteristics such as small body size, high growth rates, early reproduction and a high output of small seeds that germinate rapidly (Shipley et al.,1989; Willby et al., 2000). The early onset of flowering and seed-ripening in taxa such as *Persicaria*, *Atriplex* and *Chenopodium* can be a key characteristic of ruderals in this habitat, allowing them to complete their life-cycle within a short drawdown period. There are also some ruderal perennials including *Ranunculus repens* L, creeping bent *Agrostis stolonifera* L. and common couch *Elytrigia repens* (L.) Desv. Ex Nevski, which are strongly rhizomatous or stoloniferous and are capable of rapid vegetative spread (Grime, 1979). In a similar vein to the study by Moore et al. (1989), work on temporary ponds in the UK has shown that these sites support a disproportionately high number of rare species (Nicolet et al., 2004). It may be possible that a trend away from stable, nutrient-rich ‘core’ habitat in any direction allows scarce plants to escape competition by dominant species and this does appear to be the case in a number of different habitat types (Keddy, 2000).

Infertile, disturbed shorelines will be devoid of vegetation. The causes of this are likely to include barriers to colonization, prevention of establishment by juveniles and removal of adult plants. It has been found that biomass on lakeshores correlates positively with altitude and hence duration of exposure, showing that the frequency of disturbance is a key factor in determining plant growth (Spence, 1982; Wilson & Keddy, 1988; Gasith & Gafny, 1990).

Climate change impacts

It has been repeatedly demonstrated that the nature of the water level regime has an important influence on wetland community structure. Key factors determining species composition on shorelines are the timing, frequency and length of drawdown events, with the growing period between disturbance events being critical for marginal or terrestrial plants growing in the eulittoral zone (Meeks, 1969; Nilsson, 1979, 1981). The current situation in many lakes is for long flooded periods with short-term drawdowns in unusually dry years. This regime allows the development of submerged aquatic vegetation, with erect marginal plants at the shoreline and a fringe of wet scrub or woodland above the waterline (Spence, 1982). However, climate change is likely to result in shorter flooded periods, with increasingly frequent drawdowns of longer duration. This will have two effects: existing aquatic vegetation will be subject to stranding and a bare ground resource will become available for colonization. These processes will reduce the cover of submerged aquatics and promote the development of species adapted to the exposed eulittoral zone (Nilsson, 1979, 1981), such as ruderal mudflat annuals (Meeks, 1969; Schneider, 1994) or grasses and sedges (Gerritsen & Greening, 1989).

As an analogy for climate change impacts, a number of studies have made comparisons between regulated and non-regulated lakes, which differ in their hydroperiod regimes. Wilcox & Meeker (1991) found that an increase in annual fluctuation in a regulated lake caused a change in macrophyte communities from structurally diverse plant communities to a more limited suite of rosette and mat-forming species, the high level of disturbance reducing structural diversity. Smith et al., (1987) recorded that lakes used for hydropower, with regular large fluctuations, were devoid of littoral macrophytes, in contrast to natural lakes and water supply reservoirs. Hill et al. (1998) found that regulated lakes were less diverse, contained more exotic species and were usually devoid of rare herbs, when compared to unregulated waterbodies. Finally, Rorslett (1989), studying hydroelectric lakes, found a decline in species richness, disappearance of shallow and mid-depth vegetation and an increased incidence of ruderal plant types.

Considering these findings and CSR theory, the model in Fig. 2 indicates that, depending on the starting point of the ecosystem, one of three pathways will be induced by the hydrological changes caused by climate change.

1. Low disturbance, fertile shorelines will be subject to heightened levels of disturbance and will become increasingly ruderal in terms of their vegetation, with highly competitive species being lost. This could, however, have a potential consequence of increasing diversity as predicted by the intermediate disturbance hypothesis (Grime, 1973), and as seen in studies of shoreline vegetation (Keddy, 1983).
2. Low disturbance, infertile habitats, when subject to increased levels of water-level fluctuations, will no longer be inhabitable, resulting in loss of plant cover and increasingly bare shores. Stress-tolerant species will be lost, with negative impacts for the distinctive rare vegetation that these habitats often support (Moore et al., 1989).
3. Shorelines currently subjected to high levels of disturbance, but that remain vegetated due to fertile conditions, will perhaps become

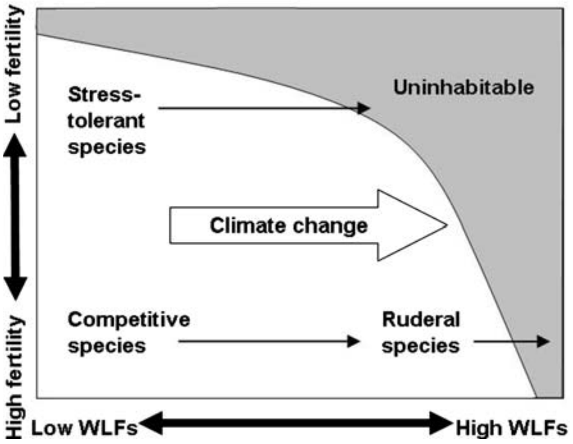

Fig. 2 Climate change impacts on shorelines

increasingly ruderal. However, there is also the potential that the frequency, duration or magnitude of hydrological changes will become too severe for even ruderal shoreline plants to cope, and plant cover will decrease, leaving bare shorelines, or those dominated by only a few species, as seen in rapid-cycling hydroelectric reservoirs (Smith et al., 1987)

In summary, the overall change in species composition in many shoreline habitats subject to increased water-level fluctuations is likely to be a loss of competitive and stress-tolerant species, with increasingly ruderal vegetation types and expanding areas of bare substrate. This will have obvious major impacts on the nature conservation value, ecosystem functioning and ecological services provided by wetland habitats. In a study of Great Lakes wetlands, Mortsch (1998) concluded that an increased frequency and duration of low water levels produced by climate change, together with changes in the timing and amplitude of seasonal water levels, would affect wildlife, waterfowl and fish habitats, water quality, wetland area and vegetation diversity. However, other evidence also suggests that, where lakes have suffered a decline in habitat or species diversity through artificial stabilization of lake levels, it is possible that climate-change induced fluctuations could reverse the adverse impacts and restore biodiversity interest (Wilcox & Meeker, 1991; Hill et al., 1998). Lakes that have become dominated by extensive stands of large competitive species could, with increased water-level fluctuations, develop a wider species complement through the creation of niches for a more diverse range of less competitive species.

Management options

Although significant levels of research have focused on climate change prediction, impact assessment and mitigation, there has been little attempt, thus far, to develop practical adaptation methods to reduce expected impacts on wetlands (Hulme, 2005). Such measures could increase the flexibility of management of important sites, enhance the possibilities for ecosystems to adapt to change and reduce the additional pressures of non-climate related impacts. A non-interventionist approach can be taken, accepting the changes to environments that will occur and allowing new habitats and communities to develop without substantial input. In many cases however, a more appropriate approach will be to implement active management strategies to mitigate the most severe effects of environmental change and allow adaptation to altered hydrologic regimes. Examples of these include the use of high priority management actions in valuable wetlands, including hydraulic controls for some wetlands to improve water management and the implementation of wetland rehabilitation and restoration projects (van Dam et al., 2002).

Building upon the potential biological effects indicated in Fig. 2, the conservation management of shorelines subject to hydrologic fluctuations and wave action should focus on four key issues (Fig. 3): hydrological management, substrate conditions, shoreline topography and the potential need for vegetation establishment (Abrahams, 2005; Abrahams, in press). These will allow the adverse impacts of increased disturbance to be mitigated and allow the protection and modification of shoreline fertility, if required, so that viable vegetation can be maintained despite the effects of climate change.

Moderate water-level fluctuations make a positive contribution to the diversity and conservation value of shoreline vegetation (Pieczynska, 1990; Schneider, 1994). However, extreme fluctuations will reduce plant cover and impoverish communities (Smith et al., 1987; Hawes et al. 2003). The threshold

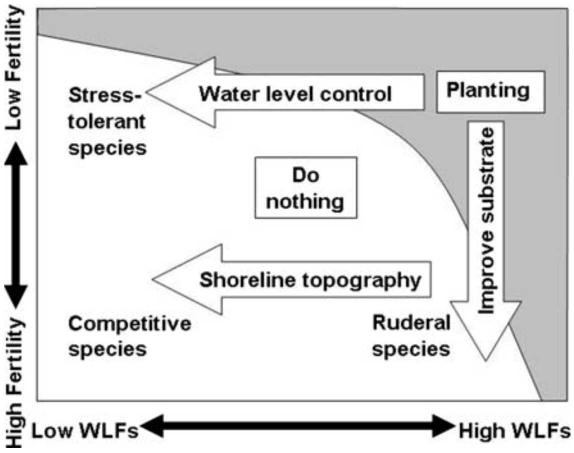

Fig. 3 Potential management options to allow adaptation to climate change

between these two levels of disturbance may be variable and is usually site-specific. There is, however, reasonable correspondence among different studies on the appropriate limits to water-level fluctuations. In temperate-zone lakes, rates of change that can sustain vegetation and produce species-rich assemblages are generally less than 0.6–2 m per month, with annual changes not exceeding 2–5 m (Petts, 1990; Wilcox & Meeker, 1991; Pyrovetsi & Papastergiadou, 1992; Smith et al., 1987). Annual fluctuations greater than 5–7 m are often too extreme and may significantly degrade littoral biota, leaving the water devoid of permanent aquatic vegetation (Nilsson, 1979; Smith et al., 1987; Rorslett, 1989; Pyrovetsi & Papastergiadou, 1992). The CSR model indicates that water level control as an adaptation strategy should seek to limit fluctuations within these thresholds, especially in lakes with nutrient-poor shorelines. For illustration, Wilcox & Meeker (1991) found that maximum species richness was produced with an annual within-year fluctuation of 1.8 m, and Hill et al. (1998) recommended an annual variation of 0.5–1 m, while Hawes et al. (2003) found the highest species richness in a range of lakes to occur in those with a monthly range of 1 m. These figures provide some indication of scale to the disturbance axis in Figs. 2 and 3.

Ecological Regulation Practices (Mark & Johnson, 1985; Hellsten et al., 1996a) and other water level management plans can offer a potential way to integrate nature conservation and operational demands in reservoirs and regulated lakes. A set of control rules can be developed to manage levels and prevent adverse ecological impacts occurring. Although there are likely to be difficulties in implementing these on sites that have high demands for water supply and flood regulation, particularly as operational demands increase, they do offer a potential method for adaptation on sites managed for nature conservation. An alternative approach to cope with increasing annual variability in the water budgets of wetland nature reserves, is to provide for additional winter water storage in connected reservoirs, so that water can be released into these systems during drier summer months (Merritt, 1994). This will enable existing hydroperiods in these systems to be maintained in their current states.

In areas of poor shoreline substrate, increased hydrological fluctuation could easily lead to the loss of capacity for plant growth as the combination of low fertility and high disturbance prevent the establishment and growth of vegetation. A potential adaptation strategy here is to reduce habitat severity by improving substrate quality (particularly organic matter content) through wave protection and substrate enhancement. The impacts of wave action, especially on a mobile sandy or silty substrate, will substantially reduce substrate organic matter content (Nilsson, 1981; Keddy, 1983). As a response to this situation, ploughing, disking and fertilizer application have all been used to improve substrate characteristics in reservoir revegetation schemes (Fowler & Hammer, 1976; Middleton, 1995). In addition, imported material can be added to the substrate to improve its suitability for plant growth, often also incorporating plant propagules to aid colonization and establishment of suitable wetland plants (Levine & Willard, 1990).

Action can also be taken to reduce the potential for wave creation, and lessen the impacts of waves on shorelines. Shelterbelts can be planted to reduce wind speed, and water depths can be reduced to decrease wave height. Artificial reefs can be built at a distance from the shoreline, generally placed in about 1 m depth of water. These installations break the waves, erosion is minimized and sheltered lagoons are created (Levine & Willard, 1990; Ferguson, 1999; Dallaire, 2001). In addition to this type of fixed barrier, the use of floating timber booms can also dampen wave energy before it reaches the shore (Andrews & Kinsman, 1990; Hall et al., 1993; Merritt, 1994). Once wave-reducing measures have been utilized, substrate support structures such as biodegradable geotextiles or other meshes provide further erosion protection for both substrate and vegetation (Cranfield University, 1999). These all help to reduce disturbance from wave impacts as well as enhancing fertility levels.

Shallow shoreline gradients will help prevent wave action, the transport of fine sediments and a consequently low fertility environment. If water levels cannot adequately be controlled in the main part of a waterbody, then stable conditions can be provided in marginal areas by the construction of bunds across bays and inlets (Reitan & Sandvik, 1996). These are used to retain a separately controlled regime or fixed depth of water in at least part of the site. Such areas may act as ‘source’ sites for emigration to the drawdown area, which may be a 'sink' for some species. On a smaller scale, the use of ridge and furrow or creation of hummocks and hollows in the drawdown zone can hold small pools of water of use to aquatic flora and fauna (Smith et al., 1987; Harper et al., 1990).

Even prior to climate change impacts, highly disturbed, low fertility severe shorelines will be devoid of vegetation. It has been hypothesized, however, that this may be due to the prevention of germination and establishment of juvenile plants (Gill & Bradshaw, 1971). If adult plants are introduced, they may be able to survive conditions that have blocked vegetation establishment, maintaining themselves despite stressed and disturbed conditions. Such artificial introductions can then help to reduce habitat severity, by introducing organic matter from leaf fall and preventing erosion of fine substrates by absorption of wave energy and the substrate binding properties of the root matrix provided (Little & Jones, 1979). Thus, revegetation schemes may be able to counteract the combined adverse effects of high disturbance and low fertility conditions.

A number of trials in different shoreline habitats have shown that a range of species are suitable for planting in areas subject to high levels of disturbance from water-level fluctuations (Little & Jones, 1979; Levine & Willard, 1990; Hellsten et al., 1996b; Fraisse et al., 1997). The use of appropriate species and management techniques, tailored to the situation and used at the correct time of year, can create plant communities that will survive and even benefit from flooding and exposure (Little, 1977; Allen & Klimas, 1986; Allen, 1988). An experimental revegetation, which showed mixed success, was undertaken by Hellsten et al. (1996b) on an eroded sandy shoreline in northern Finland. After the first summer, the average survival rates were around 45% due to the drying of seedlings, but these decreased during a high-water period until only 20% of the planted individuals were still alive. The best results were obtained with bottle sedge *Carex rostrata* Stokes, of which 30% survived and tea-leaved willows *Salix phylicifolia* L. with a survival rate of 80%. Fraisse et al. (1997) gained success in trials with eight species, but found that failures were caused by adverse hydrogical conditions or frost. Plants clearly need to have carefully controlled conditions, including water level management and quality substrate, during the early establishment phase.

Conclusion

It is anticipated that climate change will impact upon freshwater systems in a variety of ways through changes to parameters such as water temperatures, stratification processes, nutrient regime and dissolved gases (Dawson et al., 2001). However, given the fundamental importance of hydrological regimes to the functioning of aquatic systems it is likely that climate change impacts on this factor will have the most impact on sites, communities and individual species. The current paucity of research and guidance in this area is, therefore, frustrating. The biological effects of these environmental changes may include reductions in the numbers of target species, increase of invasive species, replacement of macrophyte vegetation with phytoplankton-dominated communities and an overall loss of biodiversity. These trends will prevent the effective conservation and restoration of wetlands (Mooij et al., 2005; Wei & Chow-Fraser, 2006). A greater understanding of the processes linking water-level fluctuations and littoral ecology and how these relate to climate change is a key issue that needs to be addressed in order to understand the possible compositions of future ecosystems and to develop suitable adaptive strategies to cope with climate change impacts (Gasith & Gafny, 1990; Dawson et al., 2001). Across Europe, the potential impacts of climate change on wetlands will become increasingly important with the implementation of the Water Framework Directive (Directive 2000/60/EC), which requires all waterbodies to attain 'Good Ecological Status' by 2015. The opposing forces of climate change, increasing water use and this legislation will place large demands on water resource managers, nature conservation bodies and society in general. If biodiversity, landscape and amenity elements are to be protected alongside these urgent concerns, the promotion of clear science-based management techniques will be critical (Sutherland et al., 2004). A shift in management thinking will also be required, with a move away from the paradigm that aims to ensure ecosystem stability, towards one able to predict and manage short and long-term change.

This article attempts to bring together, into one framework, some of the science relating to shoreline vegetation, the potential effects of climate change on wetlands and an introduction to practical adaptation methods. It is hoped that the use of simple models that incorporate aspects of ecology, such as the one proposed here using life history traits, will provide a valid structure for hypothesis generation and testing and will allow good habitat management decisions to be made and implemented in coming years on lakes, reservoirs and other wetlands of conservation value.

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