MITIGATION BANKING AS A TOOL FOR STRATEGIC COASTAL ZONE MANAGEMENT: A UK PERSPECTIVE

by

Stephen Crooks and Laure Ledoux

CSERGE Working Paper GEC 99-02

MITIGATION BANKING AS A TOOL FOR STRATEGIC COASTAL ZONE MANAGEMENT: A UK PERSPECTIVE

by

Stephen Crooks and Laure Ledoux

Centre for Social and Economic Research on the Global Environment
University of East Anglia
and
University College London

Acknowledgements
The Centre for Social and Economic Research on the Global Environment (CSERGE) is a designated research centre of the UK Economic and Social Research Council (ESRC).

This research paper forms part of a Natural Environment Research Council (NERC) and Environment Agency funded Research Fellowship GTS/96/ENEC/1, in association with the Plymouth Marine Laboratory and the Institute of Terrestrial Ecology. The research was also partly funded by the Economic and Social Science Research Council (ESRC). The authors gratefully acknowledge helpful comments from Sandra McNally, Kerry Turner and Tim O’Riordan.

ISSN 0967-8875

Table of contents

Abstract

Coastal zones are experiencing increasing pressure around the globe, and many countries now recognise the need for a more strategic approach to the management of natural resources. Within the no-net-loss objective of the European Union Habitats Directive, compensation for habitat displacement has become a major issue. This article examines how mitigation banking has been applied in the States in the implementation of the no-net-loss of wetlands federal policy, and how it could be used as part as a strategic management plan in the UK coastal zone. The principles of mitigation banking require that habitats be restored or recreated *in advance of* development to compensate for forthcoming displacement, thus generating credits representing the value of the restored habitat. To satisfy regulatory requirements developers withdraw debits from the bank, i.e. buy credits to compensate for the habitats displaced on the development site. This article argues that, given the feasibility of restoring certain ecosystems in the UK coastal zone, with sufficient regulation by conservation agencies and thorough scientific investigation, mitigation banking could provide a potential mechanism to compensate for incremental losses of habitat. With careful regional planning, it may also contribute to restoring and enhancing biodiversity. However, further research is required to examine how mitigation banking could be operationalised in the UK coastal zones.

1. Introduction

Coastal zones are coming under increasing pressure around the globe, from a range of driving forces, both man-made or natural. Degradation and loss of coastal natural resources are a major issue in many countries. In the face of growing land-use pressure from urban and industrial development, and the on-going decline in the amount of coastal habitats, there has been growing awareness of the need of a more strategic approach to UK coastal zone management (House of Commons, 1992).

The convention on biological diversity recognises loss of habitats as a problem with far reaching implications. In Europe, the Habitats Directive is an attempt to confront habitat and species loss, with the long-term aim of achieving an overall no-net-loss within designated areas representing European main habitats. If for reasons of overriding public interests habitats are destroyed, there must be compensation for this loss. No-net-loss is also a policy that has been implemented in the United Sates, in the context of wetland[1] protection. One instrument which has come into being and which is now common practice in the US is mitigation banking or land banking. Habitats are restored or recreated in advance of development to compensate for forthcoming displacement, and thus generate credits representing the value of the restored habitat, which can be used to compensate for future development. To satisfy regulatory requirements, developers have to withdraw debits from the bank, i.e. buy credits to compensate for the habitats displaced on the development site.

This article examines how mitigation banking could be used in the context of the Habitats Directive to achieve a no-net-loss of coastal zone habitats in the UK. The experience of wetland mitigation and the development of mitigation banking in the States are first reviewed. Economic aspects of land banking are then explored in more detail, before addressing the pros and cons of mitigation banking to achieve habitat no-net-loss. Finally, the following issues are discussed: What are the implications of the Habitats Directive for the UK coastal zone? Are coastal habitats substitutable? How do landscape considerations affect mitigation? This finally leads to a discussion about the potential of mitigation banking within coastal zone management in the UK. It is argued that strategy and long term planning are key elements to the success of such a scheme.

---

[1]Wetland is defined by the Ramsar Convention as an area of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, brackish or salt, including areas of marine waters the depth of which does not exceed six metres. This definition includes a broad range of habitats from peatlands and forested systems to subtidal sea-grasses.

2. Wetland Conservation in the United States, No-net-loss and Mitigation Requirements

No-net-loss of wetland resources has been a policy goal of both the Bush and Clinton Administrations (White House 1991, 1993). The U.S. has suffered heavily from wetland agricultural conversion and development in wetland areas. Over the period between the 1780's and 1980's total wetland area declined from 221 million acres to 104 million acres - some states losing over 80% of wetland area (Dahl, 1990). Much of this loss was due to agricultural drainage in rural areas but also to urban and industrial development particularly in coastal regions.

With growing awareness that wetlands provide valuable functions to society as well as ecological value, legislative steps were taken to initially halt and then attempt to reverse wetland decline. Early attempts by Congress to reduce wetland loss began in the 1950's with enactment of legislation, the Wetland Loan Act of 1961 and amendments to the Migratory Bird Stamp Act of 1934, primarily focused on the most visible wetland functions of habitat for waterfowl roosting and feeding. As other wetland functions became increasingly apparent further legislation was enacted. The Coastal Barrier Resource Act of 1982 put a stop to federal funding assistance to the development of coastal barrier areas, (including their wetland habitats) along the Atlantic seaboard and shores of the Great Lakes. Likewise, inland, the Swampbuster provision of the Food Security act of 1985 denied government subsidies to farmers for the drainage of agricultural land, the primary cause of wetland habitat loss.

It was with the instigation of the Clean Water Act: Section 404 [33 U.S.C. 1344 (1972)], that mitigation[2] requirements for wetland loss became a significant component of wetland conservation and formed a central part to later no-net-loss policies. Section 404 specifically regulates the placement of dredge and fill material into all waters of the United States. The term *all waters* has been interpreted in its widest sense and it is under this section of the Act that land-use change of wetlands is regulated. Under the Act, permits are requested if a development or activity is likely to have an impact on a wetland and is granted on condition that proposed activity meets the requirements stipulated under Section 404(b)(1)[3]. In most cases permitting is, by formal regulation, expected

[2] Wetland mitigation may be defined as the restoration, creation, enhancement and, in exceptional circumstances, preservation of wetlands expressly for the purpose of mitigating unavoidable losses due to authorised development actions.

[3] Permitting activity is administered by both the U.S. Corp of Engineers (Corps) and the U.S. Environmental Protection Agency (EPA), with advisory roles provided by the U.S. Fish and

to follow a hierarchical sequence of avoidance, minimisation and compensation. First the permit applicant must show that there is no ‘practicable alternative’ to the activity which would result in a less adverse impact on the wetland. If avoidance is not possible then the minimisation of disturbance must be achieved. Finally, if a permit is granted, compensation is required involving recreation of like-for-like habitat or less favourably restoration of degraded wetlands as close as possible to the permitted activity.

Although the U.S. no-net-loss policy has helped to reduce the rate of wetland loss by development, the mitigation process itself remains controversial because of the high rate of reported project failures, the low environmental value of superficially restored habitats and the poor record of project compliance with mitigation requirements (Race and Cristie, 1982; Race, 1985; Zedler, 1987; Kusler and Kentula, 1990; O’Donnell, 1988; NRC, 1992; Roberts, 1993; Gardner, 1996; Race and Fonseca, 1996). It appears that such a dismal success rate for mitigation can not be blamed totally on incomplete scientific understanding but has its roots in institutional and regulatory failure (Race and Fonseca, 1996). Such failures have included: not ensuring that proper technical decisions are undertaken; that projects are properly planned and executed; that adequate monitoring and maintenance after initial construction is undertaken and that sufficient funding are allotted to support the project (King, 1991; King and Bohlen, 1994). In many cases the promised mitigation often failed to materialise, but the problem has been compounded by regulatory organisations being unwilling or unable to take action to ensure that permittees are satisfying mitigation requirements and to punish those which do not (Gardner, 1996).

As an example of such regulatory failures, a project appraisal undertaken in Oregon found that of 58 permits issued between 1977 and 1988 which degraded 74 ha of wetland, only 42 ha of wetland were actually created (Kentula *et al.,* 1992). Likewise, in Florida, the Department of Environmental regulation (DER) found that of 199 wetland creation projects launched under 63 permits between 1985 and 1990, the success rate for tidal and freshwater wetlands was 44 % and 12%, respectively (Veltman, 1995). The DER found non-compliance to be a major cause of mitigation failure, with only 4 of the 63 permittees complying with permit requirements (Redmond, 1995).

Beyond these regulatory failures, a major ecological limitation of site-by-site mitigation is the risk and uncertainty in attempting to restore habitat functions after the original habitat has been lost. (Roberts, 1993; Zedler, 1996a). As

Wildlife Service (USFWS), National Marine Fisheries Service (NMFS) and a number of state resource agencies.

Kustler and Kentula (1990) point out in a review of restoration science, total duplication of natural wetlands is impossible due to the complexity and variations in natural systems, and the subtle relationship between hydrology, solid, vegetation, animal life, and nutrients which may have developed over thousands of years. These problems are particularly apparent in terrestrial habitats, such as mature forests and peat systems, so the use of mitigation should be tightly controlled. However - with particular relevance to coastal zone management - Kustler and Kentula go on to conclude that a relatively high degree of success has been achieved in restoring and recreating wetlands in estuarine, coastal and freshwater marshes, in that order. This higher success rate in recreating tidal wetlands, they argue, reflects the dynamic and self-regulatory nature of coastal systems. Obtaining a self-regulatory system, with no inputs of energy or material from a manager, is the key to success in ecological restoration (Jackson *et al*., 1995).

Mitigation banking seeks to address some of the shortcomings of site-by-site mitigation, and proper implementation can help to avoid some of the past mistakes; however there are still some drawbacks related to it.

3. Mitigation Banking

In this context, the mechanism of wetland mitigation banking has appeared, which involves the restoration, creation enhancement, or in except-ional cases only, preservation, of an area of functioning wetland *in advance of*, and to offset unavoidable losses due to authorised development of wetlands, preferably within the same *ecoregion.*

The aim of a mitigation bank is to reconcile two competing interests: those of developers whose actions will impact upon a habitat and those of the agency tasked with environmental conservation. Instead of requiring developers to create and maintain an environmental system, which is a role they are not skilled to perform, with sufficient regulation it may be economically and ecologically beneficial to allow them to purchase part or whole of an existing wetland, restored and maintained by a third party.

Mitigation banks may exist in a number of forms (Marsh *et al.*, 1996). Most commonly found in the U.S. are *single owner/use banks* usually set up by a large company or agency whose future development plans are anticipated to displace habitat. The company may create (usually on its own land) a single large site from which it later withdraws credits rather than undergo case-by-case iterative mitigation. Slow to emerge but now expanding rapidly with support from U.S. federal guidance and legislature are *Entrepreneurial Banks*. These are similar in principle to single user banks except that the banks are established by a third party landowner/investor and from whom credits may be purchased by developers. Under such commercial ventures, responsibility of the venture is transferred from the permit applicant to the mitigation bank owner. Currently, there are 140 banks in the U.S. with some 150 in planning stages (R. Brumbaugh,[4] 1998, personal communication). Common in California are *joint projects* whereby a number of developers may co-operate to fund a bank to compensate for anticipated further environmental impacts. Public and non-profit making ventures also exist, which may charge a fee based on the estimated cost of providing the necessary mitigation.

Mitigation banking involves the quantification of wetland functions and values as bank credits, which are either withdrawn by single users, or sold piecemeal to a number of developers to mitigate for wetland displacement. Developers have to buy enough credits to compensate for the displaced habitat in order to obtain a development permit.

[4] U.S. Army Corps of Engineers.

In the case of several users of a bank, a *mitigation credit market* develops (Scodari and Shabman, 1995). The market is defined as the relationship between the demand and the supply of credits in some geographical area. The demand side of the market is made up of consumers of wetland credits, the permittees. The supply-side of the market is made up of sellers of wetland mitigation credits, commercial or public mitigation credit ventures. As long as demanders are willing to pay a price at least equal to, or greater than the cost to supply credits, credits will be produced.

Furthermore, in theory, mitigation wetlands should be of similar kind and quality as wetlands converted by development. In practice, however, out of kind mitigation might take place, and wetlands then be of different value. In such cases, mitigation ratios should be applied. After examining issues linked to values of wetlands and costs of restoration, the process of setting the price of credits in the context of a market will be developed, along with the issue of mitigation ratios.

3.1 The value of credits

The value of credits attributed to a land bank should theoretically be based on the social value of benefits from enhancement, restoration or creation. More precisely, a mitigation credit is a unit measure of the increase in wetland value achieved at a wetland mitigation site. Mitigation credits serve as the unit of exchange for the provision of compensatory mitigation (Shabman *et al.,* 1996). Let us assume for example that benefits are monetised, and that a bank of 100 ha is created, which benefits are valued at the sum of 10 000 $. Credits will therefore be worth 100 $/ha.

Developers will have to withdraw credits from the bank (i.e. debits) to compensate for the wetland damage due to development. Debits are calculated in a similar way: if the wetland which is due to be developed is deemed to be worth 1000 $, the developer will have to buy 1000/100 = 10 ha of the land bank at a cost of 20$ each, i.e. a total cost of 200$.

The question is then how to calculate the value of the credits and the debits: i.e. how to evaluate the benefits of the converted and restored wetlands. In practice, in most of mitigation banks in the U.S., benefits derived from the land bank, or foregone benefits from developed wetlands are not calculated in monetary terms, but through biological assessment and negotiation among stakeholders.

The Federal Guidance for the Establishment, Use and Operation of Mitigation Banks (U.S. Department of Defence, 1995) specify that methods for determining credits and debits must be addressed within the banking instrument. It stipulates that “an appropriate functional assessment methodology must be used to assess wetland restoration, creation and enhancement activities within a mitigation bank, and to quantify the amount of available credits”. The guidance also states that the same methodology should be used to assess both credits and debits, so that they can both be expressed in the same units to facilitate trading, and set compensating ratios if necessary. If an appropriate functional assessment methodology is “impractical to employ, acreage may be used as a surrogate for measuring function”, but that “regardless of the method employed, the number of credits should reflect the difference between site conditions under the with- and without-bank scenarios”.

One of the biological assessment methods, suggested by the federal guidance, which provides a quantitative assessment of relative worth, with a final numeric output, is the Habitat Evaluation Procedure (HEP). It is site specific and calculates the suitability of an ecosystem as habitat through the use of a Habitat Suitability Index model for each indicator species selected. In the Anaheim Bay Mitigation Bank case (Etchart, 1995), evaluation species were chosen because they were either common to both sites or considered ecologically equivalent. The selected evaluation species included exclusively fishes and birds, but fishery resources would not be traded for avian resources, or vice versa. The 20 selected evaluation species groups, for both the landfill site and the restoration sites, together with habitat suitability indices - ranging from 0 to 1 - were determined by the judgement of each team member and averaged. Habitat unit gains and losses for the 20 listed evaluation species were compared on a unit-for-unit basis. The study concluded that for each acre constructed, about 0.759 acre of the mitigation site would compensate for the loss. In this study, the habitat before mitigation works started was considered to have no value. In general, the federal guidance specifies that “baseline values represented by existing or already planned public programs, including preservation value, should not be counted toward bank credits”.

Another example where HEP has been applied to determine the value of mitigation credits is the North Carolina Department of Transportation bank (NCDOT), established to mitigate for highway construction (McCrain, 1992). Indicator species included: grey squirrel, mink, wood duck, hairy woodpecker. Primary considerations for species selection included public visibility, economic importance, ecological importance, utilisation of representative cover types, phylogenetic representation, home range size, availability of adequate Habitat Suitability Index models, and utility for assessing impact on bottom-land hardwoods attributable to highway construction. It was found that overall, the mitigation credit value from the bank was equal to 47.9 Habitat Units per acre, taking into account the value of the habitat before restoration (i.e. after having subtracted the value of the bank before works were carried out). Based on this, it was estimated that a replacement ration of 3:1 would fully compensate for lost values on future highway projects.

3.2 The cost of credits

In cases of single user or non-profit banks, the price of credits is directly linked to the costs incurred for mitigation, but factors included as mitigation costs and the resulting fee calculation vary across the mitigation banking experiences in the States. Mitigation costs to be taken into consideration in the calculation of the fee may include costs related to: hiring of wetland restoration or creation experts, planning (e.g. site selection), land acquisition, project implementation (restoration, creation or enhancement), site management, monitoring, administration (Apogee research, 1993, Scodari and Shabman, 1995).

As well as the cash costs listed above, there might also be opportunity costs. The opportunity cost of land, for example, is the forgone benefit that could have been made in the next best alternative use, or the forgone sales value. If restoration takes place on valuable agricultural land, the opportunity cost is likely be high. Subtleties in defining costs arise from the use of inputs that are donated to, or already owned by, the credit venture, but cannot be sold. For example, government ventures might use lands that they already own, and that are dedicated to wildlife habitat, and other compatible uses for credit production. If this land is required by law to be held in public trust for perpetuity, there is no opportunity cost to credit suppliers.

Furthermore, because of concerns about the success of mitigation, in some cases, regulatory agencies establish a set of trading rules, including for example the requirement that credit producers post a performance bond. The agency may hold the entire amount of the bond until the mitigation project is judged completely successful, or it may release portions of the bond when certain milestones are satisfied. Performance bonds can also be used to ensure long-term maintenance of mitigation sites (Gardner, 1996). Cash costs from the assurance bond then derive from portions of the bond not being returned, or repair costs incurred to earn the return of the full bond amount in the event of site failure. Associated opportunity costs are the interest charges on the cash value of the bond until its reimbursement by the regulatory agency (Scodari and Shabman, 1995). The use of performance bonds is often linked to the possibility of selling credits before the success of the bank can be ascertained: if sales are allowed early, banks will usually be required to post a performance bond. If sales are only allowed when the wetland is successfully restored, performance bonds are deemed not to be necessary.

In early projects, direct restoration costs only were considered. Monitoring and long-term management costs were rarely included in compensatory fees. Exceptions include the Pine flatwood wetlands mitigation trust, in Louisiana, where compensation must provide the trust fund with sufficient resources to cover inventory, acquisition, management, and administrative costs of the mitigation program, over a 50 years period. The way mitigation fees are calculated seem to depend, from the US experience, on whether the mitigation bank is a public body or a private enterprise. In the early days, private entrepreneurs seemed to ignore certain costs, such as monitoring, to bring the credit prices down and be more attractive to developers, whereas the perception was that public bodies were more concerned about the true costs of quality restoration and long term monitoring (King, 1991, King and Bohlen, 1994). Scodari and Shabman (1995) report a different picture. In their analysis, allowing for credits sales prior to the attainment of performance standards in return for provisions requiring the ventures to post financial assurances for the success of replacement wetlands has improved the quality of private entrepreneurs' mitigation banks. On the other hand, the authors state two cases of public commercial ventures which do not ensure quality controls necessary for ecological success, as they were not required to post financial assurances, even though they were authorised to sell credits before replacement wetlands were demonstrated to be successful. In these cases, ventures initially fell short of mitigation goals and required significant remedial action. It is clear that whether the venture is private or public, success rates tend to be linked to the true estimation of costs, in particular to the provision of funds for long-term management.

Fees can be based on average costs, or vary over time or space to reflect increasing marginal costs. For example, in Dade County, Florida, fees were based on the estimated cost per acre of enhancement in the East Everglades, and increased from 1989 to 1993 to reflect the change in *Melaleuca* eradication costs as the programme progressed. As enhancement work moved farther away from the Park, the density of *Melaleuca* increased, requiring more efforts in terms of mechanical and chemical treatment. On the contrary, The Ohio wetlands foundation determined the per acre mitigation fee, by calculating the expected total cost of design and construction at the selected mitigation site, and dividing this amount by the total acreage (Apogee Research, 1993).

The cost of restoration itself will obviously depend on the type of habitat which is to be recreated or restored, and on initial geomorphological conditions. Some coastal ecosystems are easier, and therefore less costly to recreate. King and Bohlen (1994) analysed costs associated with nearly 1000 agricultural conversion projects and mitigation projects and obtained average costs for nine types of wetland restoration projects (Table 1)[5].

Table 1: Average cost per acre for different wetland types

Wetland type	Project Average cost (thousands of $)	Sample size
Aquatic beds - tidal or non-tidal communities of permanently or nearly permanently submerged plants	9.5	3
Complex projects incorporating three or more wetland types in a single project	56.7	8
Freshwater mixed projects, consisting of non-tidal projects in which both forested and emergent vegetation is produced	25.3	10
Freshwater, non-tidal projects establishing forested wetlands	77.9	19
Freshwater, non-tidal projects establishing emergent wetlands	48.7	28
Projects producing tidal freshwater wetlands	42	3
Projects establishing salt marshes and other marine or estuarine wetlands dominated by emergent vegetation	18.1	9
Projects establishing mangrove communities	18	4
Agricultural conversions	1	494

Source: King and Bohlen, 1994.

The authors first conclude that wetland restoration undertaken as part of voluntary programmes in which agricultural lands are converted to wetlands are substantially less costly, on average, than other mitigation projects. These projects are generally fairly simple operations, and usually involve restoring original site hydrology (e.g. by breaking drainage tiles or filling ditches), which

[5] Cost estimates include pre-construction, construction, and post-construction costs, but exclude land costs.

is inexpensive and often highly successful. Agricultural conversion projects also involve fewer restoration tasks than restoring structurally and biologically more complex wetlands, and therefore are much less expensive. In contrast, for the rest of the categories, the authors found that differences in wetland type had a surprisingly small effect on average restoration costs once site specific and project-specific factors were accounted for. In fact, project costs varied more widely within wetland categories than between wetland categories. It is interesting to note, however, that coastal ecosystems such as salt marshes and other estuarine wetlands are among the lower cost restoration projects.

The cost will also vary with the objectives of the bank. In Sweden for example, a series of wetlands were created in the context of a programme aimed at encouraging creation of wetland for nitrogen reduction. The costs involved in the creation of 53 created wetlands were analysed, and preliminary results show that the principal factors explaining the variation of costs were area, and construction efforts, in particular the need for excavation (Söderqvist, 1998). These wetlands were created in the context of a programme aimed at encouraging creation of wetland for nitrogen reduction, so costs involved are probably a lower estimate to wetland creation costs since other aspects such as recreation and biodiversity were not considered.

Costs are also related to the time scale of restoration, and the likelihood of success. As investment in the restoration project increases, the ecosystem is likely to recover more quickly, and the likelihood of success increases, although this is only true within a range of values. In other words, both the level of recovery and the speed of recovery, depend on how much is spent on the restoration effort (King, 1991). In general, because success can never be guaranteed, the relevant concept is probability of success, which may be a function of site characteristics, funding available or long-term maintenance of site.

3.3 Credit markets, price of credits and mitigation ratios

If the bank owner has no profit-making objective, then the price of credits will just be set to reflect the costs of mitigation. In the case of banks created by private entrepreneurs, however, the bank owners will seek to maximise profits from selling credits, and the price will be established according to demand and supply. The demand for venture mitigation credits is a function of: overall development pressure, the relative return from development on wetlands compared to uplands, the expectation of receiving a wetland development permit, the costs of mitigation undertaken by the permittees relative to that from the purchase of venture credits, and regulatory permission to deviate from the sequencing requirements to use on-site mitigation. Credit supply is a function of the costs that ventures incur when producing credits, which themselves is a function of trading rules in place (e.g. performance bonds) (Scodari and Shabman, 1995).

The total market is supplied by ventures with different cost structures. Some ventures will earn higher net returns than others, for example because they have a unique skill (restoration expertise) or asset quality (location of mitigation land). All ventures will expect to recover "commercial" costs of production. These are the costs that the venture deems relevant to the attainment of its financial objective, which might be to earn costs plus a small mark-up, or to break-even. Commercial costs of earning credits are not necessarily comparable across different ventures, but are specific to the circumstances of a particular venture. For example, low average unit cost government ventures might be due to the government not considering land or management as commercial production costs. Furthermore, government ventures often have a break-even financial objective. The market will therefore be dominated by ventures (i) that have cost advantages, (ii) that have accounting perspectives, and/or make accounting judgements that do not consider certain expenses to be costs, and/or (iii) that have a financial goal other than to maximise net returns. Shabman *et al.*, (1996) underline that if the price-setting process for public banks does not reflect all bank costs, public banks will in effect directly subsidise mitigation of permit applicants, and introduce "below-cost" competition for private banks. But there might also be particular circumstances where efficiencies, for example scale economies, or lower failure risk costs, would justify a lower public price than private price.

The concept of trading ratio, as introduced by Shabman *et al.*, (1996), also directly influences the price of credits. Once the value of credits has been established, a credit trading ratio different from 1:1 might be established by regulators because of additional reasons: the trading ratio might be adjusted upward to account for the risk of mitigation failure. Regulators may also want to adjust trading ratios upward because of possible temporal losses in wetland functioning. Finally, regulators may want to adjust trading ratios to ensure that bank trades result in no net loss in wetland acreage as well as function, if the replacement wetlands are judged to have greater ecological value than the impacted wetlands.

Once the credit ratio is established in a particular situation, developers will bid for credits. The overall price of a development permit for a particular company will be the credit market price, times the number of credits necessary to satisfy the compensation requirements. The price of a credit will be established somewhere between the bank owner's cost of producing a credit and the developer's maximum willingness to pay for it (equal to the profit of development per credit gained), depending on the market conditions. If the bank is the only one in the region, the owner is likely to be able to extract much of the surplus from development. On the contrary, if there is high competition between banks, the price of credits is likely to be lower. Shabman *et al.*, (1996) suggest interestingly that there might be a third case: before setting the trading ratio. The regulator might know the permit applicant's willingness to pay, and the credit supplier's minimum acceptable price, and correspondingly establish a higher trading ratio, so that most of the development surplus is transferred to the restoration of wetland.

It is worth underlining, along with Shabman *et al.*, (1996) that the supply and demand conditions in markets for mitigation credits are exceptional because of the two roles that must be played by the government. First, in the absence of regulations creating the demand for development permits, mitigation markets would not exist. Second, permit applicants are price-conscious, but not quality conscious. Their only concern is that they satisfy the requirements for compensatory mitigation. It is the regulator who must impose quality control through trading rules.

When values per acre between developed and mitigation sites differ, in either direction, a mitigation ratio must be adopted. If the wetland impacted is deemed to be of higher ecological value than the one which is being restored in the context of the land bank, the ratio will be more than one: to offset development of one acre of wetland, more than one acre of land bank has to be bought in compensation. In the scope of the 1987 Chesapeake Bay Agreement, a mitigation bank was established, and the Maryland Department of Natural Resources (DNR) accepted fee-based compensation for mitigation requirements if it determined that creation, restoration, or enhancement of non-tidal wetlands was not feasible. DNR determined the mitigation acreage requirements as a function of the size of the permitted impact and an established mitigation ratio - 3:1, 2:1, or 1:1 per acre. Similarly, in the Pine flatwood wetlands mitigation trust, in Louisiana, the amount of compensation was determined through a standardised process ("Ecological Value Assessment") that aimed to quantify the overall natural quality of the pine wetlands in the impacted area (Apogee Research, 1993).

4. The Pros and Cons of Mitigation Banking

4.1 Benefits of mitigation banking

An advantage of an up and running mitigation bank over direct site-by-site compensation is that the suitable substitution for the wetlands being displaced will have been created before loss of the site, or in the process of development. This allows clear identification of whether the mitigation provided for the development meets the criteria of a like-for-like substitution or if like-for-like substitution is not possible, provides the means by which out of kind compensation might be negotiated. Many environmentalists support mitigation banking, over site-by-site projects, because credits are only granted when restoration and creation efforts are judged successful, thus preventing situations where wetlands are destroyed, and the ensuing mitigation effort fails (Veltman, 1995). Reciprocally, when mitigation takes place in advance, and if the initiators of mitigation banks estimate thoroughly the costs of mitigation, banks provide developers with a predictable measure of costs involved, and allow them to make an economically rational decision based on full costs and benefits for development. In the case of site-by-site mitigation, it is unclear to a developer prior to creation of a site-by-site mitigation whether it will be successful. If not then further unbudgeted resources will be required to meet the restoration objectives.

Having a mitigation bank in existence also reduces the time lag between the loss of the existing habitat and the development of the mitigated wetland. This may be a considerable problem for site-by-site mitigation where a considerable temporal lag exists between displaced and recreated wetlands. This time lag will depend on the complexity of the wetland form and function and may for some habitats take upward of several decades to be achieved.

A large mitigation bank may provide ecological benefits outweighing those provided by a large number of small, non-contiguous wetland mitigation. In site-by-site mitigation, the set-aside or restored land may satisfy the legal requirements, but may not itself be of high ecological importance, or may not be connected to other protected open space with priority habitat value, and over the long term the set-aside land may lose some or all of its biological value because of its small size. Overall, isolated mitigation projects that have little connection with their surrounding ecosystem often are more prone to failure than a mitigation project that is incorporated into a larger, ecosystem-based conservation bank or regional conservation plan. Off site mitigation provides a greater selection of hydrologically and ecologically favourable locations, thus increasing the opportunity for a well-functioning replacement (Veltman, 1995, Kusler, 1992). Larger systems are also more self-sustaining because they can provide habitat for more types of species, more sustaining food chains, which in turn can better accommodate ecosystem succession, migration and changes (Kusler, 1992).

The costs of participating in a mitigation bank may be considerably cheaper than creating a number of smaller individual mitigation sites. Land banks provide economies of scale relating to the planning, implementation, monitoring and management of mitigation projects, and all administrative costs. It also allows pooling scientific resources and expertise together, providing better advice at a lesser cost. Turner and Boyer (1997) seem to argue, in the case of creating new coastal wetlands by river diversion in Louisiana, that there are diseconomies of scale, i.e. that smaller projects are more efficient than others. It is difficult to generalise this result, however, for several reasons: the costs included in this analysis are not clearly detailed. The bigger projects have additional aims of maintaining a predictable and stable river environment for flood protection and navigation, compared to the smaller ones. This requires significant additional costs that increase exponentially with project size. The authors also underline that smaller diversions will create wetlands at slower rates than larger projects, at least over several decades. King and Bohlen (1994) studied the costs of 1000 wetland restoration projects and concluded that the costs of mitigation for larger projects were lower on a per-acre basis than the costs for smaller projects, comparing the same type of mitigation projects. This is likely to be true in general, as significant fixed costs are associated with all but the most simple restoration projects, and there can be significant quantity discounts and labour cost savings as the size of projects increases.

If the sites are properly valued, and if regulators ensure quality matters in the trade-off, when the owner of the bank produces higher-quality wetlands, the bank's credits will result in more favourable compensation ratios for the bank owner, allowing for potentially higher bank profits. A quality-conscious incentive will exist for the bank owner to recreate high-quality wetlands (Gilman, 1997; King and Bohlen, 1994). This incentive did not exist in the site-by-site mitigation framework, and has contributed to poor quality wetland restoration.

In the U.S. site-by-site mitigation, because of its high failure rate, has been attributed with contributing to the cumulative loss of coastal wetlands. The existence of an 'up-and-running' mitigation bank would, on the one hand limit this high failure rate, and on the other hand, facilitate substitution for small areas of land, and might strengthen arguments by conservation organisations that compensation for even the smallest development be provided.

In the US experience, project-by-project mitigation often involves lengthy regulatory processes and significant costs for private landowners seeking project approvals. Conservation banks can greatly ease these burdens, reducing mitigation compliance to a single transaction and giving project participants the certainty of having complied with mandated mitigation requirements. Mitigation banks serve to streamline in general the regulatory process for those parties needing to mitigate for projects, and thus simplify the regulatory compliance process. Provisions for monitoring mitigated wetlands' long term viability and consequences for failure are also more easily enforced under a banking system (Veltman, 1995).

4.2 Limitations

There are still limitations regarding mitigation banking.

A potential problem is that permitting authorities do not enforce the proper sequencing of the project: avoidance, minimisation, and compensation, i.e. that credits are distributed only when wetland destruction is unavoidable. Allowing destruction of more wetlands than strictly necessary means that the number of wetland acres that must be restored and created increases, which in turn increases the chances for failed mitigation (Veltman, 1995).

When mitigation does take place in advance of development, one of the problems is finding the initial funds to set up the bank, since credits can usually only be sold once the mitigation project is successful. In the US, the federal guidance allows for a limited withdrawal of credits in the bank's early stages. However, no early credits can be used unless: the Mitigation Bank review team has approved the banking instrument and mitigation plan, the bank sponsor has obtained the mitigation site, and the banking instrument contains "appropriate financial assurances" (Gardner, 1996). In Florida, to provide an incentive for bankers to create mitigation banks, and to move away from site-by-site mitigation, the regulations allow for the release of credits in blocks based on the attainment of specific performance criteria (Redmond, 1995). For example 15% of credits might be released when a land management arrangement has been conveyed; 20% when the grading is complete; 20% when the planting is completed; 20% at a later time interval when progress towards functional reinstatement has been shown and the final allotment when total success is attained. The relative percentages released at given milestones will vary depending on the nature of habitats being restored, with a greater quantity being withheld for more problematic (but higher credit value) wetlands until final ecological success has been demonstrated.

Because of incomplete scientific knowledge of the functions of wetland systems, allowing degradation of complex wetlands in exchange for compensatory mitigation might result in a loss of unknown and irreplaceable wetland functions. It was recently argued (Mitsch and Wilson, 1996) that a relatively high number of failures of mitigation wetlands could be attributed to a general lack of understanding of first principles of wetland science. The authors incriminate in particular little understanding of wetland function by those constructing the wetlands, insufficient time for the wetlands to develop, and a lack of recognition or underestimation of the self-design capacity of nature.

Furthermore, certain wetland functions are site specific and cannot be satisfactorily replaced by mitigation banking. Mitsch (1998) underlines that a lot of wetland values are dependent on where they are found in the landscape, for example, the degree to which a wetland is open to hydrologic and biological fluxes with other systems. Another example is the difference of value a wetland will have for nutrient removal, whether it is upstream or downstream. Similarly, the efficiency of wetlands in decreasing flooding increases with the distance the wetland is downstream. This explains why off-site mitigation may be felt to be an unacceptable alternative to on-site habitat restoration by conservation organisations. It is therefore important that sufficient investigation of the proposed displaced wetland is undertaken to determine the functions and ecological character.

Finally, if there is no overall strategy between mitigation banks across a region, and if out-of-kind mitigation is allowed without planning, land banking may result in loss of wetland diversity. Bedford (1996) argues that as the number of exchanges of one ecosystem for another increases, mitigation changes from a regulatory action to a landscape policy, with the potential to reduce the diversity of wetland ecosystem types, and warns about the potential cumulative effects of individual mitigation projects on broad scale patterns of wetland diversity. Land banking can address this problem, to the condition that criteria by which to judge replacement are established, and that a regional strategy is set up to maintain diversity. Zedler (1996b) suggests, in the context of Californian coastal wetlands, that a suitable regional priority would be re-establishment of habitats that have been most reduced in area and species, and that it is more important to retain and expand on the habitat remnants, rather than add new ecosystem types.

This also depends on the proper evaluation of habitats, both those to be developed, and the ones restored for compensation. The most commonly used evaluation procedure, and one recommended by the US federal guidance - the Habitat Evaluation Procedure - has been judged to be “technically defensible, replicable, and applicable in a variety of different habitat types" (McGrain, 1992), but it has also been widely criticised by ecologists and economists alike. From the ecologists' point of view, it relies only on species, and does not take into account other functions of habitats, such as nutrient removal, or flood control. From the economists' point of view, it is not based on people's preferences. If the overall no-net-loss objective, and more particularly, no loss of diversity, is to be achieved, a full economic assessment such as cost benefit analysis will not ensure that wetland diversity is maintained. On the other hand, it is important that a consistent framework is established to compare the costs and benefits of restoration alternatives, and as King (1991) argues, there is no such framework at the moment. A useful step forward would be to deal explicitly with both the cost and the expected results of alternatives, rather than relying on subjective performance measures and imperfect information. King (ibid.) also suggests that since projects will be aimed at the restoration of several different ecological functions, more than one set of performance measures may be required to evaluate and compare the alternatives.

5. Mitigation Banking and Coastal Zone Management in the UK

5.1 The Habitats Directive and mitigation requirements

In Europe, obligatory compensation for displacement of designated habitat within protected sites is a recent concept, driven by E.U. legislation, specifically the Habitats Directive[6]. This Directive also encompasses the previously existing Birds Directive[7] and together they provide a legislative framework intended to protect wildlife and habitat designated as of European importance (DG XI.D, 1996). As such, the Habitats Directive which has subsequently been adopted into UK law by SI.2716 (Conservation Regulations) in 1994, is the single most important piece of conservation legislation to influence the UK.

The Habitats Directive is effectively a no-net-loss policy enacted to provide protection to species and habitats from development pressures, within designated Natura 2000 sites[8]. These sites are to be made up of existing Special Protection Area (SPAs), to conserve the 182 bird species and sub-species listed under annex I of the Birds Directive as well as migratory birds, and of new Special Areas of Conservation (SACs), to conserve 253 habitat types, 200 animal and 434 plant species listed under annex I and Annex II of the Habitats Directive.

Under the Directive, member states are required to maintain habitat and species levels at a *favourable conservation status,* defined in terms of the natural range being stable or increasing and requiring the existence of structures and functions necessary for the long term maintenance of that site[9]. At present the UK holds 140 SPA and an incomplete list of 262 candidate SAC sites representing conservation coverage of 5,046 km^2 and 15, 268 km^2 respectively. Within the coastal environment so far 12 candidate marine SACs have been proposed, and many "terrestrial" habitats are located in the coastal fringe

---

6. Council Directive 92/43/EEC on the conservation of natural habitat and wild flora and fauna (known as the Habitats Directive), adopted in May 1992.

7 Council Directive 79/49/EEC on the protection of wild birds, adopted in 1979.

8 Under Article 6(2) *Member States shall take appropriate steps to avoid, in SACs, the deterioration of natural habitats and the habitat species as well as disturbance of species for which the area has been designated, in so far as such disturbance could be defined significant in relation to the Directive.*

9 Habitat site selection (methodology outlined in Annex III of the Habitats Directive) is based on representatives of habitat type, area of habitat in relation to national total coverage and ecological quality (including restoration potential). Similarly, for individual species candidate SACs are identified on the basis of size and density of population in relation to national population, quality of the site (including restoration possibilities) and the degree of isolation of the species on the site relative to the national population range.

(dunes, mud flats, coastal freshwater wetlands, etc.)

Articles 6(3) and 6(4) aim to ensure that the Natura 2000 site is not damaged by a new *plan or project* before the implications of which are given full consideration balancing the nature conservation and opposing interests. If it is found that the plan or project will adversely affect the integrity of the site, the competent authority may approve it only under certain conditions: 1) it must be clear that there are *no alternative solutions*; 2) the plan or project must represent an *overriding public interest* and 3) the Member State must adopt *compensatory measures* which may include habitat restoration or recreation of same habitat type on the same site or elsewhere. An additional dimension is that where a site hosts a priority natural habitat and/or a priority species, an over-riding interest justifying a project must relate to human health or safety, or be of primary importance for the environment, or further to an opinion of the EC, relate to some other imperative reason.

Given the current pressures in the UK coastal zones from industry, tourism, and port development, and given the number of coastal designated areas, mitigation is likely to become a major issue in coastal zone management in the coming years.

5.2 Are coastal habitats substitutable?

The concept of no-net-loss and mitigation is founded around the belief that the habitat functions are substitutable - that those habitats lost can be restored or recreated elsewhere. As we have seen from research in the U.S., identical replacement of wetlands is, by and large, not possible because of the complexity of ecological systems. Nevertheless it has also been found, in both the U.S. and in Europe that some success has been achieved in restoring or recreating certain forms of coastal habitat.

Coastal floodplains support a diverse range of habitats, including: intertidal flats, saltmarshes, sand dunes and dune slacks, shingle structures, saline lagoons, wet grasslands, freshwater lakes, marsh ditches and reed beds, peat-lands (including bog and fens), carr and ancient woodlands. There is some experience in the UK in creating and restoring a number of coastal systems. Much of the wet grasslands habitats found in the coastal zone (current national levels: 220,000 ha) are artificial in origin, created by embankment of coastal and riverine floodplains (Benstead *et al.*, 1997). Large areas of land are estimated to provide suitable opportunities for the restoration of wet grasslands (perhaps some 1,200,000 ha in all) and with correct land and water management it is possible to reinstate many of these habitats within a period of less than ten years (*ibid*). Nature reserves have been highly successful in creating reedbeds, freshwater ponds and brackish-saline lagoons, primarily for birds, over a similar time period (P. Doktor[10], 1998, personal communication). By and large, however, these habitats require management to control water levels and salinities, so objectives of creating self-sustaining hydrology are not generally attained.

In the intertidal zone, recent managed retreat (realignment) experiments and historic 'natural' storm breaches in sea-defences show that it is possible to restore mudflat and saltmarsh habitats (IECS, 1992; Pye and French, 1993; Environment Agency, 1998; Reading *et al.*, 1998). The degree to which the form and function of the restored saltmarsh habitat mirrors that of neighbouring marshes is, however, regionally variable, depending on a range of factors (i.e. site history and management, surface elevation and topography, tidal characteristics, nature of substrate) (Crooks and Pye forthcoming). Other habitats are more difficult to restore. Dunes and shingle structures may be enhanced on site and managed with increase sensitivity to optimise environmental benefits. However, being highly sensitive to sediment supply it is not feasibly possible to recreate these habitats as part of an off-site mitigation beyond allowing existing habitats to migrate landward from existing positions across back-barrier lands. Habitats such as peatlands, carr and ancient woodlands can also be recognised as habitats which require considerable periods of time to mature (10^2-10^3 years), if possible at all.

Just because an 'immature' form of habitat can be restored does not mean that all such habitats are therefore substitutable. The conservational and functional value of any habitat to be displaced must be carefully considered as mature habitats may perform functions which the younger habitat might not be capable of (i.e. support of endangered species or complex ecological community structures). In attempting to provide some clarification on which habitats are or are not irreplaceable English Nature (1994, 1995a,b) have adopted the approach of dividing habitats into Critical Natural Capital (CNC) and Constant Natural Assets (CNA). CNC are defined as 'those elements of the natural environment '....which would be *irreplaceable*, or which would be too difficult or expensive to replace in human time scales whilst CNA are 'those elements of the environment which individually can be sacrificed for development as long as compensatory action of equal worth is taken'. It is intended that these definitions run in parallel to established statutory and non-statutory site designations. Identifying CNC is based on whether the site has any rare or declining species, supports characteristic species assemblages for a habitat type, provides important environmental services and the length of time required to

[10] Norfolk Wildlife Trust.

substitute the habitat, if at all possible. In line with principles of sustainable development and intergenerational equity, if a habitat form is not replaceable within 25 years - taken to reflect one generation - then its is classified as CNC.

5.3 Landscape considerations in siting a mitigation bank

Subject to environmental change (rising sea-level, climatic warming, etc.), the most suitable place to restore a habitat is on a site at which it once existed. Across the UK, there are some 615,000 ha (National Farmers Union, 1998) of embanked coastal floodplain lying beneath the level of tidal waters. Although some of this has been developed, the remainder offers possible opportunities for some form of habitat restoration, particularly with declining requirements for agricultural production.

When considering siting a mitigation bank it is important not only to be aware of site issues (former land use, hydrology, substrate, topography etc.) but also to place its functioning of the desired restoration in the wider landscape context. A number of questions should be asked. What will be the effect of any coastal realignment on the morphodynamics of the estuary? Are ecological linkages and functional benefits maximised? Will the project further ecoregion biodiversity management goals? How is land use expected to change and will this affect the future integrity of the site?

Restoring habitat in the intertidal zone, by realigning or opening up flood defences, will have implications for the tidal prism of an estuary. Generally, estuaries are widest at the mouth and taper towards the head. In response to this funnelling effect, the progressing tidal wave becomes steeper and with reduced frictional drag travels faster in to the estuary than the subsequent ebbing tide drains out. As net particle flux is very sensitive to the balance between flood and ebb tide velocities (Dronkers, 1986), estuaries are generally sinks for sediment. Embankment in many cases has altered this natural shape, influenced tidal dynamics and sedimentation patterns. Pethick (1993, 1994) argues that in planning to manage an estuary sustainably for the future, the effect of moving embankments on tidal hydraulics should be carefully modelled with the long-term goal of returning estuarine morphology to a more natural shape. Opening up defences in the inner estuary will increase tidal storage capacity and the increased through flow of water will cause an enlargement of the cross-sectional area at the mouth, and erosion of outer estuarine intertidal habitats. Additionally, enlargement of inner estuary may, in an ebb-dominated system, exacerbate tidal erosion, so possibly leading to an estuarine-wide loss of habitat. Retreat of embankments in the outer estuary may have the opposite effect and in time lead to a more stable estuarine form. By the same rationale, further embankment may decrease tidal volume in the inner estuary and, along with restriction of estuarine channel migration and flushing, lead to enhanced sedimentation in flood dominated systems (e.g. Pye and Neal, 1994). This strategy in the long run though, will lead to a net decline in intertidal and estuarine subtidal levels.

Heterogeneity in landscape structure over varying spatial and temporal scales influences the way organism, populations, and whole assemblages interact with the environment (With and Crist, 1995; Gustafson and Gardener, 1996). With-in coastal fisheries, ecosystem heterogeneity may enhance biodiversity by supporting recruitment and maintaining the numbers of species which require multiple resources (Parrish, 1989; Irlandi and Crawford, 1997; Primavera, 1998). In addition, landscape composition and structural heterogeneity act to influence biological interactions of competition and predation (Coen *et al*., 1981; Danielson, 1991), foraging behaviour, and predator avoidance (Mittelbach, 1986; Werner and Hall, 1988; Cattrijsse *et al*., 1997). Considering salt marshes, Broome *et al.,* (1988) recognised the influence of nearby sand dune complexes on water retention in restored marshes. Sacco *et al.,* (1994) and Moy and Leven (1991) described how the proximity of natural marshes to a restored salt marsh aided the development of faunal communities. Similarly, poor halophyte establishment of a restored marsh in southwest England was interpreted by Boorman (1997, personal communication) to reflect the scarcity of salt marshes and lack of propagules within the region. By studying pinfish (*Lagodon* rhomboides), Irlandi and Crawford (1997) identified functional link-ages between salt marsh and sea grass habitats; recognising that pinfish productivity was linked to landscape heterogeneity. Wetlands are also known to have greater biodiversity value if linkages are maintained with adjacent upland ecosystems. Enhancement of overall estuarine ecological functioning requires that restoration projects be planned incorporating landscape element heterogeneity, connectivity and the flux of material to and from their surroundings (Bell *et al*., 1997; Ehrenfeld and Toth, 1997). However, to date, because of poor understanding of these linkages, restoration objectives have rarely been considered beyond a site limited context. A number of steps might be considered to maximise the contribution of a restoration project to ecological functioning: creation of habitats which are known to be a limiting factor in local ecological functioning; restoration of multiple habitat forms on large projects (maybe dependant on topographical constraints); and siting of restoration project to maximise ecological linkages.

Drawing conservation plans and setting biodiversity goals on a hierarchy of geographical scales from international to local level is now common practice (Bibby, 1998), though the consideration of which scale to base a restoration strategy on, and issues of habitat relocation can be contentious. To maintain biodiversity and ecosystem integrity requires planning that extends beyond individual sites but considers the wider regional biogeography and landscape patterns, perhaps even above local concerns (Noss, 1983). In support of this, Kupfer (1995) highlights the importance of conservation planners to recognise the significance of large-scale processes. First it promotes a shift towards large-scale population dynamics within spatially heterogeneous landscapes. Individual habitats should not be considered as self-contained entities, rather the effects of external forces and inter-ecosystem linkages need to be recognised and incorporated in to conservation action plans. Secondly, it stresses that conservation efforts should be made with greater awareness of effects of both spatial and temporal scales. It therefore becomes important for restoration efforts to move away from focusing on small-scale piecemeal issues towards that of an integrated approach, particularly in fragmented landscapes (Zedler, 1996b).

Also to be considered is whether restoration should be restricted to replacing habitat which is being lost from the contemporary landscape, or seek to reintroduce habitats which once existed historically. A common characteristic of many estuaries is the embankment and loss of freshwater tidal and upper intertidal habitats leaving a fringe of lower intertidal habitats. Given sufficient evaluation, it may be that some flexibility is required within no-net-loss objectives to allow previously displaced habitats and their functions to be restored as part of any compensatory package to offset any projected losses of the existing intertidal ecosystems.

Finally, future land use changes may impact on the restoration site (e.g. by causing disturbance, changes to material flux, disruption to wildlife corridors, etc.) and where possible these should be taken into account within a management strategy. In addition, any site used to provide compensatory habitat should be provided with adequate protection from future development. A term entering the American vocabulary on restoration is ‘wandering mitigation’ describing the repeated movement of a single mitigation site to offset sequential development mitigation protects. Taking a theoretical example to help clarify this concept, consider the series of urban developments in a coastal area. Initially a permission might be sought to displace a wetland for which a nearby restoration project is required and undertaken. Later, a subsequent development project requests development at the restoration site and offers to offset the wetland loss with equivalent wetland restoration elsewhere. Unless additional mitigation requirements are demanded by the regulator to take in to account the losses of the first project, then a net loss of habitat and functions will occur. Such a situation may perpetuate such that a number of developments are undertaken and with only a single restoration site being provided as compensation.

5.4 Mitigation banking in the UK coastal zone

Given that restoration of certain coastal habitats is technically feasible, mitigation banking can be a useful framework to implement such mitigation projects, avoiding the factors which lead to a poor success rate in the US, and in particular ensuring proper regulation by conservation bodies.

Section 404 of the Clean Water Act provides some protection to habitats which are not within national parks or under statute (i.e. Endangered Species Act) or non-statute designation (Ramsar convention). The Habitats Directive in Europe has the same objective, but only applies to a network of designated area. Although mitigation banking has been applied mostly to wetland conservation in the U.S., new initiatives have shown that it can be extended to other habitats. Mitigation banks which would restore, create or preserve coastal habitats, and from which credits would be sold to those who must compensate for resource impacts on the coastal zone elsewhere, could effectively address the objectives of the Habitats Directive in Europe.

The US experience can help avoid mistakes. First of all, it was recognised that the main factor of failure in the early experiments was regulatory failure: the principle of precedence was not respected, and there was not enough quality control. For mitigation banking to be successful in the UK, there needs to be stricter control of permits and different stages of development of land banks. One useful lesson from the States is also that performance insurance is a good way to ensure a certain level of quality of the replacement wetland, when the sale of credits is allowed before completion of the project. This also allows more flexibility to the bank as it generates cash flows earlier.

Another issue is the risk of an overall loss of habitat diversity if out of kind mitigation is allowed, i.e. if a mitigation ratio is used to allow a greater surface of a lower quality wetland to compensate for the destruction of a high quality wetland, as this might lead in the long term to regression or disappearance of certain types of ecosystems, in favour of more easily recreatable habitats, such as mudflats. This doesn't necessarily mean that out-of-kind compensation should not be allowed, but it should only take place in the context of an overall restoration strategy, identifying what is the long term balance of habitats to be achieved regionally. Objectives will vary from region to region, depending on which habitats are most threatened. In the North Norfolk coast for example, the priority is protection or recreation of freshwater reedbeds and grazing marshes, given the pressure from natural coastal erosion.

Furthermore, the US experience is interesting in looking at respective success rates of private enterprise or public ventures. The question will be similar in the UK: should the Environment agency be in charge of land banks, or conservation bodies such as English Nature, or RSPB, or private entrepreneurs? There is no definite conclusion from the US experience: in the early days, mitigation banking by private entrepreneurs was criticised because credits were underpriced to increase demand, but later experience showed that private banks could lead to successful restoration work, when performance insurance was required.

Mitigation banking can bring similar benefits as in the US experience: restoration or recreation of habitats prior to their destruction, i.e. reduced time lag between the loss of habitat and its replacement; greater ecological benefits than one-to-one small scale mitigation sites, economies of scale. But there would also be benefits specific to the context of the UK coastal zone.

Small scale but cumulative losses through development and land-use change are probably the greatest threat to coastal habitats (beyond sea-level rise and coastal squeeze). Under the Habitats Directive, compensation for habitat loss is only required if the development plan or project is deemed to have an adverse effect on the integrity of the Natura 2000 site. Because the definition of an adverse effect is not clearly defined, and because consideration of plans and projects takes place on an individual basis, small habitat loss often takes place without the requirement for compensation. Over the long-term this may lead to progressive deterioration of the site. While site-by-site mitigation is not practical for very small scale development because of fixed costs, and ecological feasibility, withdrawing a small number of credits from a bank is relatively straightforward. Mitigation banking can therefore be a useful tool to avoid this piecemeal destruction by allowing substitution for small scale impacts.

Through the selling of credits, mitigation banking provides funds available for restoration projects. This is particularly interesting given the lack of funding available for habitat restoration in the UK, especially in the coastal zone. Until recently government grant supported schemes to maintain flood defence requirements were the main funding mechanism to recreate habitat in the coastal zone, through the landward realignment of sea-defences. MAFF[11] introduced a new PGAS[12] scheme, designed to channel funds where most critical: protecting human lives and the higher economic value areas, such as urban conurbations over rural areas. Implementation of this scheme will cause

[11] Ministry of Agriculture, Fisheries and Forestry.
[12] Priorities for Grant Aid Scheme

the level of protection to rural areas to steadily fall, placing natural assets behind flood defences under increasing pressure. However, because PGAS considers projects on an individual basis and funding is geared towards urgency, long term pressures on coastal resilience are not taken into account. Although there is scope within the scheme to include intertidal habitat recreation as a benefit, both for environmental and 'soft-engineering' flood defence considerations, large-scale geographical concerns are not taken into account (Turner *et al.*, in press).

Thus, mitigation banking would provide a useful source of funding. It in fact provides a double dividend comparable to those derived from ecological taxes: on the one hand, the requirement that developers buy credits internalises the social costs linked to development - developers are faced with the full cost of development - and on the other hand, the revenues from selling credits are effectively used to restore habitats elsewhere, and thus compensate for the lack of market for coastal ecosystems and wetland benefits, which are mostly external (Turner and Jones, 1990). In the case of commercial ventures, profits will be made over and above the actual cost of mitigation, which will provide additional revenues that can potentially be used again to correct for the distortion of social under-investment in wetlands.

This would provide a "new" source of funding when there are developers who are clearly responsible for displacement of habitats. However, in certain cases, natural processes such as coastal erosion can also be an important factor in habitat loss. This is for example the case in the North Norfolk Coast, where, because of strict planning regulations, erosion is the primary threat to coastal habitats. In this context, it is difficult to determine who should be responsible for replacement. Even if anthropogenic activities can be considered to contribute to sea level rise and coastal erosion, the responsibility is shared amongst a multiplicity of actors. One solution would be to redirect existing funding sources to buying credits from land banks, which would still display significant benefits, such as economies of scale and strategic planning. Redirection of funds could for example involve agricultural subsidies, flood defence monies. In the recent House of Commons Select Committee investigation, WWF-UK reported a proposed funding to upgrade flood defences of a stretch of river in Sussex at a cost of £10 million. They argued that this money would have provided enough funding for 50 years' salt marsh restoration payments for every farmer in the floodplain. These sorts of options for the redirection of funding should be considered.

Alternatively, it is conceivable that a public body could make profits from selling credits to developers who need to compensate for their activities elsewhere. In this case, the revenues above the costs of mitigation could be used to recreate coastal wetlands and compensate for natural losses.

These funds, from developers or from redirected sources, can be used for different restoration strategies. One possibility is to use them to provide incentives for small-scale individual private projects such as restoring marginal agricultural land in farms along the coast, which presently suffer from lack of funding. Another solution is to use funding for large-scale restoration projects and strategic planning. Banks make it possible for mitigation requirements from development projects across a region to be bundled and applied at a single, high-priority site. They can be used in a strategic way to achieve regional environmental objectives. For example, land banking may become part of a strategic flood defence and management retreat programme, so reducing the costs. Banks can also serve as a major funding component for the creation of an ecosystem conservation scheme under a regional conservation plan.

6. Conclusions and Future Research

The experience of wetland mitigation in the U.S. indicated that wetland creation is problematic. To maintain wetland habitat and natural functions non-displacement should be the primary objective. However, where development is found to be necessary, mitigation in coastal areas offers a higher success rate than for terrestrial systems. Although the Habitats Directive aims at a no-net-loss of habitats, development for projects of "overriding public interest" is author-ised. Land banking can play an important role in regulating compensation for development, and provides a supporting mechanism to maintain coastal habitat levels whilst allowing economic development. A clear lesson from the U.S. is that firm regulatory procedures should be in place to ensure that: the compen-sation is provided and is of suitable quality; that only habitat considered to be a constant natural asset (i.e. substitutable) is displaced and that critical environ-mental capital (non-substitutable) is provided a greater level of protection. To ensure that the banks' ecological objectives are achieved, it is recommended to require that the bank provide some sort of performance insurance, particularly in the case of private entrepreneurs. The challenge is to set rules for credit trading which limit the risk of mitigation failure, while ensuring that they do not become cost-prohibitive, in order to preserve the economic viability of credit market systems.

Coastal zones seem to be the best place to start applying mitigation banking type of instruments in Europe: given the high natural and man-made pressures on the coast, the demand for mitigation is likely to be high, and it seems that restoration success is higher for coastal habitats. With sufficient regulation by conservation agencies and thorough scientific investigation, the authors believe that mitigation banking may provide a means to compensate for incremental losses of habitat. With careful regional planning, it may also restore and enhance habitat diversity. However, beyond this preliminary study, more research needs to be done to examine how it could be specifically implemented in the UK coastal zones.

Further research is specifically required to the costs and benefits of restoring different forms of habitat. This will need to be supported by increased under-standing and valuation of habitat functions, particularly with respect to linkages between landscape elements. Further work is necessary to clarify credit values based on functional assessment. This should be extended beyond conservation or flood defence requirements but also to include other important, but often less prominent, functions such as, for instance, benefits to the fisheries industry or water quality amelioration. Research is required to clarify the validity and possible use of mitigation ratios as part of out-of-kind mitigation. From an institutional perspective the question of who should run a mitigation bank (Government, NGO or private investor) needs to be addressed along with an evaluation of possible funding instruments. Finally, a broader, global review is required of existing no-net-loss policies to determine their effectiveness in ecosystem protection.

References

Apogee Research (1993) Alternative mechanisms for compensatory mitigation: case studies and lessons about fee-based compensatory wetlands mitigation. Working Paper, 81p.

Bedford, B.L. (1996) The need to define hydrologic equivalence at the landscape scale for fresh-water wetland mitigation. *Ecological Applications*, 6(1):57-68.

Bell, S.S., Fonseca, M.S. and Motten, L.B. (1997). Linking restoration and landscape ecology. *Restoration Ecology*, 5: 318-323.

Benstead, P., Drake, M., Jóse, P., Mountford, O., Newbold, C. and Treweek, J. (1997) *The Wet Grassland Guide: Managing Floodplain and Coastal Wet Grasslands for Wildlife.* R.S.P.B., Sandy England. 254pp.

Bibby, C.J. (1998) Selecting areas for conservation. In: W.J. Sutherland (ed.) *Conservation Science and Action*. Blackwell Science: Oxford..

Broome, S.W., Seneca, E.D. and Woodhouse Jr., W.W.. (1988) Tidal salt marsh restoration. *Aquatic Botany*, 32: 1-22.

Bower, B.T. and Turner, R.K. (1998) Characterising and analysing benefits from integrated coastal management (ICM). *Ocean and Coastal Management*, 38: 41-66.

Cattijsse, A., Dankwa, H.R., and Mees, J. (1997) Nursery function of an estuarine tidal marsh for brown shrimp *Crangon crangon*, *Journal of Sea Research*, 38: 109-121

Crooks and Pye (forthcoming) Sedimentological controls on the erosional loss of saltmarshes: Implications for habitat recreation by managed retreat In: K. Pye and J.R.L. Allen (eds) (1999) Coastal and Estuarine Environments- Sedimentology, Geomoprhology and Geoarchaeology. Geological Society Special Publication. Geological Society Publishing House, Bath.

Dahl, T.E. (1990) *Wetland Losses in the United States, 1780's to 1980's*. USID, U.S. Fish and Wildlife Service. Washington DC, 21pp.

Dronkers, J. (1986) Tidal asymmetry and estuarine morphology, *Netherlands Journal of Sea Research*, 20: 117-131.

Ehrenfeld, J.G. and Toth, L.A. (1997). Restoration ecology and the ecosytem perspective. *Restoration Ecology*, 5: 307-317.

Environment Agency (1998) *Results of Post Breach Monitoring of Orplands Coastal Realignment Site: August 1997 to March 1998.* Environment Agency, Bristol.

Etchart, G. (1995) Mitigation banks: a strategy for sustainable development. *Coastal Management*, 23: 223-237.

Gardner, R.C. (1996) Banking on entrepreneurs. *Iowa Law Review*, 81: 527-587.

Gilman, E.L. (1997) A method to investigate wetland mitigation banking for Saipan, Commonwealth of the Northern Mariana Islands. *Ocean and Coastal management*, 34(2): 117-152.

Gustafson, E.J., and Gardener, R.H. (1996). The effect of landscape heterogeneity on probability of patch colonisation. *Ecology*, 77: 94-107.

House of Commons (1992) *Coastal Zone Protection and Planning*. (Environment Committee Second ReportO HMSO, London

IECS (1992) *Historical study of sites of natural sea wall failures in Essex*, Institute of Estuarine and Coastal Studies, University of Hull.

Irlandi, E.A. and Crawford, M.K. (1997). Habitat linkages: the effect of intertidal saltmarshes and adjacent subtidal habitats on abundance, movement, and growth of an estuarine fish. *Oecologia* 110: 222-230.

Jackson, L.L., Lopoukhine, N. and Hillyard, D. (1995) Ecological restoration: a definition and coments. *Restoration Ecology*, 3: 71-76.

King, D. (1991) Costing out restoration. *Restoration and Management Notes*, 9(1): 15-21.

King, D. and Bohlen, C. (1994) Estimating the Costs of Restoration. *National Wetlands Newsletter*, 16(3): 3-5, 8.

Kentula, M.E., Brooks, R.P., Gwin, S.E., Holland, C.C., Serman, A.D. and Sifneos, J.C. (1992) An approach to improving decision making in wetland restoration and creation. In: A. J. Hairston (ed.), U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Kupfer, J.A. (1995). landscape ecology and biogeography. *Progress in Physical Geography*, 19: 18-34.

Kusler,J.A. (1992) Mitigation Banks and the replacement of wetland functions and values. In: *Effective Mitigation Mitigation Banks and Joint Projects in the Context of Wetland Mitigation Plans*. Proceedings from a National Symposium. Association of Wetland Managers, Palm Beach Gardens, Florida, 1992, pp 51-56.

Kustler, J.A. and Kentula, M.E. (eds.) (1990) *Wetland Creation and Restoration: The Status of the Science*. Washington DC, Island Press, 591pp.

Marsh, L.L., Porter, D.R. and Salvesen, D.A. (1996) *Mitigation Banking: Theory and Practice.*Island Press, Washington D.C., 300pp.

McCrain, G.R. (1992) Habitat Evaluation Procedures (HEP) applied to mitigation banking in North-Carolina. *Journal of Environmental Management*, 35(2): 153-162.

Mittelbach, G.G. (1986) Preditor-mediated habitat use: some consequences for species interactions. *Environ. Biol. Fish*. 16: 159-169.

Mitsch, W.J. (1998) The value of wetlands: the importance of landscape setting and scale. In Soderqvist, T., *Wetlands: Landscape and Institutional Perspectives*, Proceedings of the 4th Workshop of the Global Wetlands Economics Network, Beijer Occasional Paper, pp 17-30.

Mitsch, W.J., Wilson, R.F. (1996) Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecological Applications*, 6(1): 77-83.

Moy, L.D. and Leven, L.A. (1991) Are *Spartina* marshes a replaceable resource? A functional pproch to evaluation of marsh creation effort. *Estuaries,* 14: 1-16.

National Farmers Union (1998). Memorandum included in minutes of the House of Commons Agricultural Committee sixth report on Flood and Coastal Defence (Vol. II). HMSO, London, 142-155.

Natural Resource Council (1992) *Restoration of Aquatic Ecosystems: The Status of Science, Technology and Public Policy.* National Academic Press, Washington, D.C. U.S.A.

Noss (1983) A regional landscape approach to maintain diversity. *Bioscience*, 33:700-706.

O'Donnell, A. (1988) The policy implications of wetland. In: J. Zelanzy and J.S. Feierbbend (Eds.) *Increasing our wetland resources*: proceedings of a conference. National Wildlife Foundation. Washington D.C., U..S.A.

Parrish, J.D. (1989) Fish communities of inter-acting shallow-water habitats in tropical oceanic regions. *Mar. Ecol. Prog. Ser*. 58: 143-160.

Pethick, J.S. (1993). Shoreline adjustment and coastal management: physical and biological processes under accelerated sea-level rise. *The Geographyical Journal*, 159: 162-168.

Pethick, J.S. (1994) Managing managed retreat. In: A. Parker (Ed.) *Environmental Sedimentology: the interaction of water sediment and waste*, University of Reading.

Primavera, J.H. (1998). Mangroves as nurseries: shrimp populations in mangrove and non-mangrove habitat. *Estuarine, Coastal and Shelf Science,* 46: 457-464.

Pye, K. & French, P.W. (1993) *Erosion and accretion processes on British saltmarshes*, final report to the Ministry of Agriculture Fisheries and Food. Vols 1-5, Cambridge Environmental Research Consultants, Cambridge.

Pye, K. and Neal, A. (1994) Coastal dune erosion at Formby Point, north Merseyside, England: causes and mechanisms, *Marine Geology*, 119: 39-56.

Race, M.S. (1985) Critique of present wetland mitigation policies in the United States based on analysis of past restoration projects in San Fransico Bay. *Environmental Management*, 9: 71-82.

Race, M.S. and Cristie, D.R. (1982) Coastal zone development: mitigation, marsh creation and decision-making, *Enviromental Mangement*, 6: 317-328.

Race, M.S. and Fonseca, M.S. (1996) Fixing compensatory mitigation - What will it take? *Ecological Applications*, 6(1): 94-101.

Reading, C.J., Paramor, O.A.L., Garbutt, R.A., Watts, C.W., Spearman, J.R., Baratt, D.R., Cox, R., Hughes, R.G., Longstaff, D.J., Myhill, D.G., Rothery, P., Gray, A.J., (1998) *Managed Realignment at Tollesbury and Saltram*. MAFF/NERC Project Annual Report for 1997, Institute of Terrestrial Ecology (Natural Environment Research council).

Redmond, A.M. (1995) Florida moves to mitigation banking. *National Wetlands Newsletter* (Nov-Dec) 14 -17.

Roberts, L. (1993) Wetland trading is a losers game, say ecologist, *Science*, 260, 1890-1892.

Sacco, J.N., Seneca, E.D. and Wentworth, T.R.. (1994) Infaunal community development of artificially established salt marshes in North Corolina. *Estuaries*, 17: 489-500.

Scodari, P. and Shabman, L. (1995) *National Wetland Mitigation Banking Study, Commercial Wetland Mitigation Credit Markets: Theory and Practice*. Institute for Water Resources Report 95-WMB-7, 89p.

Shabman, L.A., Scodari, P., and King, D. (1996) Wetland mitigation banking markets. In: Marsh, L.L., Porter, D.R. and Salvesen, D.A. (1996) *Mitigation Banking: Theory and Practice.* Island Press, Washington D.C., 300pp.

Söderqvist, T. (1998) Empirical cost equations for wetland creation: the case of wetlands as nitrogen sinks in Scania, South Sweden. Draft paper presented at the ECOWET workshop, Norwich, 4-6 June 1998.

Turner, R.E. and Boyer, M.E. (1997) Mississippi River diversions, coastal wetland restoration/creation and an economy of scale. *Ecological Engineering*, 8:117-128.

Turner, K. and Jones, T. (1991) *Wetlands, Market and Intervention Failures. Four Case Studies*. Earthscan Pub Ltd, London.

Turner, R.K., Adger, W. N. , Crooks, S., Lorenzoni, I., Ledoux, L. (in press) Sustainable Coastal Resources Management: Principles and Practice. *Natural Resources Forum*.

U.S. Department of Defense, Environmental Protection Agency, Department of Interior, Department of Agriculture, Department of Commerce (1995). Federal Guidance for the Establishment, Use and Operation of Mitigation Banks. *Federal Register*, 60(43): 12292, March 6.

Veltman, V.C. (1995) Banking on the future of wetlands using federal law. *Northwestern University Law Review*, 89(2): 654-689.

Werner, E.E. and Hall, D.J. (1988). Onto genetic habitat shifts in bluegill: the foraging rate-preditor risk trade-off. *Ecology*, 69: 1352-1366 .

White House Office of the Press Secretary (1991) *Fact Sheet: Protecting America's Wetlands* Press release dated 9/8/1991

White House Office on Environmental Policy (1993) *Protecting America's Wetlands: A Fair, Flexible and Effective Approach.* 24/8/1993.

With, K.A. and Crist, T.O. (1995). Critical thresholds in species response to landscape structure. *Ecology*. 76: 2447-2459.

Zedler, J.B. (1987) Mitigation problem on the Southern California coast, *California Waterfront Frontage*, 3(1): 32-33.

Zedler a, J.B. (1996) Coastal mitigation in Southern California: the need for a regional restoration strategy. *Ecological Applications*, 6(1): 84-93.

Zedler b, J.B. (1996) Ecological issues in wetland mitigation - an introduction to the forum. *Ecological Applications,* 6(1): 33-37.