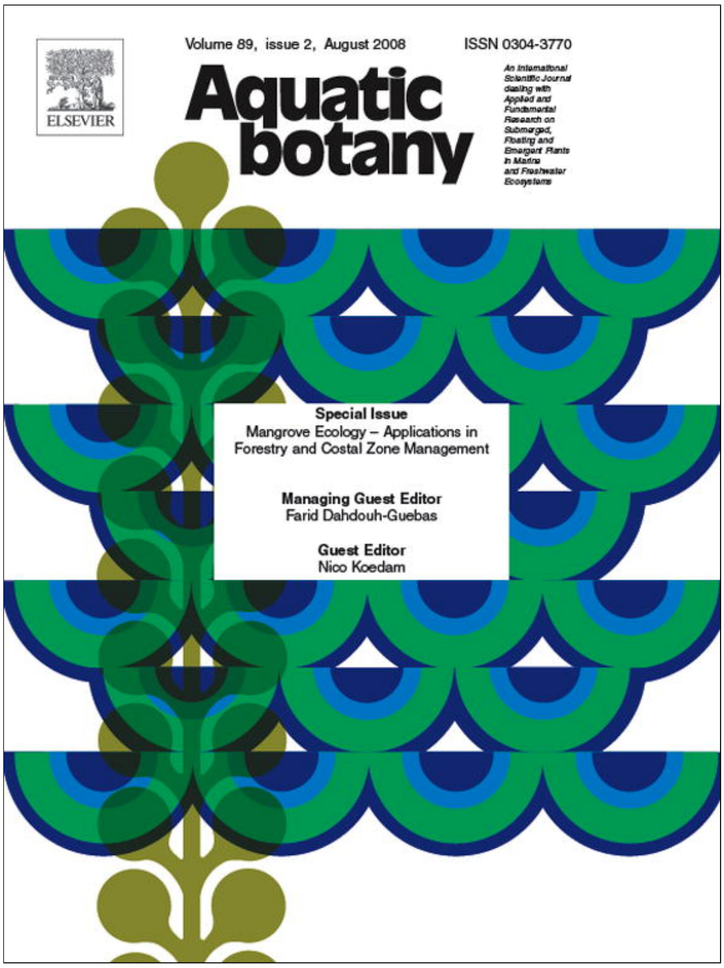
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Review

Functionality of restored mangroves: A review

J.O. Bosire [a,*], F. Dahdouh-Guebas [b,c], M. Walton [d], B.I. Crona [e], R.R. Lewis III[f], C. Field [g], J.G. Kairo [a], N. Koedam [b]

[a] Kenya Marine and Fisheries Research Institute (KMFRI), P.O. Box 81651, Mombasa, Kenya
[b] Biocomplexity Research Focus c/o Laboratory of Plant Biology and Nature Management, Mangrove Management Group, Vrije Universiteit Brussel – VUB, Pleinlaan 2, B-1050 Brussels, Belgium
[c] Biocomplexity Research Focus (Complexité et Dynamique des Systèmes Tropicaux), Département de Biologie des Organismes, Université Libre de Bruxelles – ULB, Campus du Solbosch, CP 169, Avenue Franklin D. Roosevelt 50, B-1050 Bruxelles, Belgium
[d] School of Ocean Sciences, University of Wales, Bangor, Gwynedd, Wales, UK
[e] Department of Systems Ecology, Stockholm University, S-106 91 Stockholm, Sweden
[f] Lewis Environmental Services, Inc., P.O. Box 5430, Salt Springs, FL 32134-5430, United States
[g] University of Technology, Sydney, 11 Darnley Street, Gordon, NSW 2072, Australia

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ABSTRACT

Widespread mangrove degradation coupled with the increasing awareness of the importance of these coastal forests have spurred many attempts to restore mangroves but without concomitant assessment of recovery (or otherwise) at the ecosystem level in many areas. This paper reviews literature on the recovery of restored mangrove ecosystems using relevant functional indicators. While stand structure in mangrove stands is dependent on age, site conditions and silvicultural management, published data indicates that stem densities are higher in restored mangroves than comparable natural stands; the converse is true for basal area. Biomass increment rates have been found to be higher in younger stands than older stands (e.g. 12 t ha^{-1} year^{-1} for a 12 years plantation compared to 5.1 t ha^{-1} year^{-1} for a 80-year-old plantation). Disparities in patterns of tree species recruitment into the restored stands have been observed with some stands having linear recruitment rates with time (hence enhancing stand complexity), while some older stands completely lacked the understorey. Biodiversity assessments suggest that some fauna species are more responsive to mangrove degradation (e.g. herbivorous crabs and mollusks in general), and thus mangrove restoration encourages the return of such species, in some cases to levels equivalent to those in comparable natural stands. The paper finally recommends various mangrove restoration pathways in a functional framework dependent on site conditions and emphasizes community involvement and ecosystem level monitoring as integral components of restoration projects.

Contents

* Corresponding author. Tel.: +254 11 475154; fax: +254 11 475157.
E-mail address: bosire98@yahoo.com (J.O. Bosire).

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Table 1
Yield table data for mangrove plantations at Gazi

Parameters	Utilization classes (cm)				
	<4.0	4.1–6.0	6.1–9.0	9.1–13	Total
Stems ha^{-1}	559	1586	2392	327	4864
Merchantable volume[a] (m^3 ha^{-1})	1.56	11.63	37.81	9.7	60.71
Un-merchantable volume (m^3 ha^{-1})					43.09
Standing biomass (t ha^{-1})	2.35	18.55	66.36	19.39	106.66
Below ground biomass (t ha^{-1})					24.89

[a] Volume equation used is y, $4a^2 \times 10^{-10} + 3a \times 10^{-5} + 2a \times 10^{-5}$, where y is the stem volume, and $a = D_{130}^2 H$ (*source*: Kairo et al., 2008).

1. Introduction

Towards the end of the twentieth century, scientific concern began to focus on the unprecedented loss of naturally occurring mangroves ecosystems around the world (Walsh et al., 1975). In 1983, UNDP and UNESCO established a regional project concerned with the value of mangrove ecosystems in Asia and the Pacific.

This international initiative led to an increased appreciation of the value of mangroves and a subsequent upsurge of mangrove restoration efforts (Field, 1996; Kairo et al., 2001). Some of the objectives driving early mangrove reforestation efforts include: wood production for timber, poles and fuel wood; fisheries productivity; coastal protection against storms, and legislative compliance (Ong, 1982; Field, 1996; Saenger, 2002). The rationale for mangrove restoration has changed very slowly over the years from just silviculture to recognition of mangroves as a diverse resource. The term 'restore' is taken to mean the creation of a sustainable functioning mangrove ecosystem that may or may not resemble its precursor at the very same site.

The early attempts at mangrove restoration met with mixed results with some being successful, while others were doomed from the start (Field, 1996; Erftemeijer and Lewis, 1999). Most of these attempts were not based on well-understood ecological principles and well-defined aims.

In more recent times, attention has turned to the ecological processes present in natural and restored mangrove systems (Alongi, 2002; Saenger, 2002; Lewis, 2005; McKee and Faulkner, 2000). The relationship between the restored mangrove ecosystem and adjoining ecosystems, such as salt marsh (Santilan and Hasimoto, 1999) and seagrass beds (Hogarth, 2007) has also been a focus of attention. A consensus has emerged that an understanding of mangrove hydrology is most important for successful restoration (Wolanski et al., 1992).

Ellison (2000) did a comprehensive review on mangrove restoration examining goals of existing restoration projects, and whether these goals address the full range of biological diversity and ecological processes of mangrove ecosystems. He pointed out that the focus on silviculture remained the primary objective of mangrove restoration and that few species had been involved and indicated that adequate data exists to enable successful mangrove restoration but emphasized that assessment of structural and functional characteristics of restored mangroves is imperative. This paper takes Ellison's review (Ellison, 2000) further and presents a comprehensive review of the data available on the functionality of restored mangrove ecosystems in respect to a number of functional indicators: vegetation structure, natural regeneration, productivity, nutrient recycling to conservation of inherent biodiversity and socio-economic valuation. Finally, it looks at the constraints and opportunities for successful mangrove restoration. Within the context of this review, functionality is used to refer to the ability of restored mangroves to process nutrients and organic matter, trap sediments, provide food and habitat for animals, protect shorelines, provide plant products and a scenic environment, in a similar fashion to natural mangrove forests. These aspects are often referred to as the goods and services that mangroves can provide (Walters et al., 2008).

2. Forest structure, biomass and regeneration

2.1. Structure, regeneration and biomass of restored mangroves

Most of the studies on mangrove forest structure and regeneration have focused on natural stands (e.g. Cole et al., 1999; Kairo et al., 2002); with relatively few references on reforested stands such as in the Matang forest reserve (Putz and Chan, 1986; Ong et al., 1995); as well as Ranong in Thailand (FAO, 1985; Choudhury, 1997) and Sundarban in India (Hussain, 1995; Choudhury, 1997). Apart from studies by Bosire et al. (2003, 2006), and Kairo et al. (2008), at Gazi bay in Kenya, little is known about structural development of replanted mangroves in Africa.

Analysis of stand table data from a 12 years old (Table 1) *Rhizophora mucronata* Lamk plantation in Kenya indicate that reforested plots have the potential of yielding 4864 stems ha^{-1} (much higher than the stem density in a natural stand of the same species at the same site of 1796 stems ha^{-1}; Bosire et al., 2006), with a standing biomass and merchantable volume of 106.7 t ha^{-1} and 60.7 m^3 ha^{-1}, respectively (Kairo et al., 2008). This standing biomass is much lower than the 240 t ha^{-1} observed in a nearby *R. mucronata* natural stand (Slim et al., 1997). The root biomass value in replanted *R. mucronata* was 24.9 ± 11.4 t ha^{-1}; which is 19% of the total plant biomass (Kairo et al., 2008). A review of literature on biomass studies indicates that root biomass values vary from one study to another depending on the method used (e.g. Vogt et al., 1998) and the data obtained in Kenya is comparable to ranges observed for *Rhizophora* studies in Thailand (Alongi and Dixon, 2000).

The biomass accumulation rate for the 12-year-old *Rhizophora* plantation in Kenya was estimated at 12 t ha^{-1} $year^{-1}$ (Kairo et al., 2008). This value is higher than the 5.1 t ha^{-1} $year^{-1}$ reported for an 80-year-old natural plantation of *Rhizophora apiculata* Bl. in Malaysia (Putz and Chan, 1986). In Matang mangrove forest, Ong et al. (1995) reported aboveground biomass increment of 24.5 t ha^{-1} $year^{-1}$ (and 34 t ha^{-1} $year^{-1}$ when belowground biomass was included) for 20-year-old plantation. It is logical to conclude that biomass accumulation rate is influenced by age, species, management system applied, as well climate.

The mean canopy height for the 12-year-old *Rhizophora* plantation in Kenya was 8.4 ± 1.1 m (range: 3.0–11.0 m) with a mean stem diameter of 6.2 ± 1.9 cm (range: 2.5–12.4 cm). These values are within the range reported for *Rhizophora* plantations in South East Asia (see, e.g. Srivasatava et al., 1988; FAO, 1994). Based on growth data, the mean annual increment (MAI) in height and diameter (DBH) for the *Rhizophora* plantation in Kenya were 0.71 m and 0.53 cm, respectively. These figures are within the range of published mangrove growth rates (7–12 m for height, and 5–15 cm for diameter) in Asia and Pacific (Watson, 1931; Durant, 1941; Putz and Chan, 1986; UNDP/UNESCO, 1991; Devoe and Cole, 1998;

Saenger, 2002). The basal area for 12-year-old *R. mucronata* was 16.5 m^2 ha^{-1}, which was lower than that of a natural stand of the same species (e.g. Bosire et al., 2006). This is expected since, despite having a higher stand density than a natural stand, most of the stems were of smaller size classes. A decline in stand density and an increase in basal area are typical for a developing forest (Twilley, 1995).

2.2. Composition and pattern of natural regeneration

Seedling recruitment and survivorship principally drives population growth (Burns and Ogden, 1985; Krauss et al., 2008) and thus determines the quality of the crop and productivity of forest stands (Srivastava and Bal, 1984). This becomes even more critical in restored mangrove sites where for economic reasons, many plantations tend to be monocultures (Walters, 2000; Bosire et al., 2006) Therefore evaluation of the regeneration potential of a stand, in terms of seedling density, composition, sizes and the possibility of recruitment into the adult canopy.

When conducting natural regeneration sampling in mangroves, newly recruited juveniles measuring 30 cm and below are referred to as 'potential regeneration. Individuals greater than 30 cm and higher are termed 'established regeneration, whereas those greater than 150 cm are saplings or young trees. For adequate natural regeneration a minimum of 2500 well-distributed seedlings per hectare has been proposed (Srivastava and Bal, 1984).

The recruitment rate of saplings has been found to be increasing with age in one *R. mucronata* Lamk. plantation in Kenya (Fig. 1). The densities observed in this plantation are however, much lower than those observed in a comparable conspecific natural stand at the same location (see, e.g. Kairo et al., 2002; Bosire et al., 2006), suggesting age may be a critical factor in determining the level of natural regeneration. In subsequent assessments, the canopy species has dominated the juvenile density in contrast to *Bruguiera gymnorrhiza* (L.) Lamk. dominance at earlier stages of forest development (Bosire et al., 2003). Some non-planted mangrove species have also been recruited into the adult canopy of the same plantation hence enhancing stand complexity (Bosire et al., 2003, 2006) contrary to a *S. alba* replanted stand of the same age where species encountered as juveniles experienced 100% mortality and thus none were observed in the adult canopy. This mortality of non-conspecific species was attributed to tidal submergence and barnacle infestation typical of this inundation class (Bosire et al., 2006). Contrary to observations in Kenya, Walters (2000) found no post-planting sapling recruitment in 50–60-year-old *R. mucronata* plantations in the Philippines probably due to periodic removal (weeding) of non-planted species by locals and in some plantations no actual natural colonization at all.

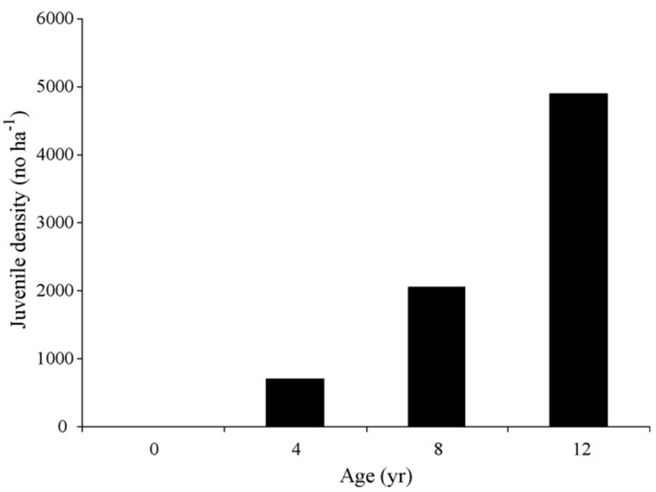

Fig. 1. Sapling recruitment over time in a *R. mucronata* plantation in Kenya.

3. Biodiversity in restored mangroves

While mangrove associated fauna play such a significant role in the functioning of the ecosystem (Kristensen, 2007; Lee, 2007; Cannicci et al., 2008; Kristensen et al., 2008; Nagelkerken et al., 2008) and thus can be a useful indicator of the state of managed mangroves, silvicultural management more often than not ignores assessing this component (Ellison, 2007). This section will highlight some trends in recolonization of epibiotic, macrobenthic and sediment-infauna communities and also look at distribution patterns for benthic macrofauna, fish and shrimp in replanted stands across the world, with focus on species richness and community assemblages.

3.1. Epibiotic and epibenthic communities

It is important to investigate to what extent mangrove restoration does support faunal recolonisation. In Thailand, crab diversity at some of the replanted sites was higher than at an upper shore natural mangrove site, and both biomass and crab numbers were consistently higher in the replanted sites (Macintosh et al., 2002). However, the natural site was characterized by large numbers of sesarmid crabs. Differences in the crab diversity in Thailand were reported to relate to inundation zone and differences in the mangrove species present in the replanted sites (Macintosh et al., 2002). However, in Qatar (Al-Khayat and Jones, 1999) found lower species richness of crabs in plantations compared to natural habitats of *Avicennia marina* (Forssk.) Vierh.

In Kenya, reforested stands of *R. mucronata* and *A. marina* had higher crab densities than their natural references (Bosire et al., 2004) but with similar species diversity and crab species composition compared to bare controls with similar site history. More sesarmid species were observed in the reforested stands (similar to the natural references) than the bare controls. Since sesarmids are thought to be key stone species with respect to nutrient recycling (Kristensen et al., 2008; Cannicci et al., 2008), they therefore seem more responsive to ecosystem degradation or restoration. In the Philippines the relative abundance of the exploited mud crab *Scylla olivacea* (Herbst) compared to two other non-commercial species was used to separate the effects of habitat from fishing pressure and recruitment limitation. A comparison of mud crab populations in replanted, natural and degraded sites in the Philippines suggested that 16 years old replanted *Rhizophora* spp. can support densities of mud crabs equivalent to that of natural mixed species mangroves (Walton et al., 2007).

Mollusc diversity showed similar patterns to that of crabs in both previously mentioned studies in Qatar and Thailand, while in Kenya, no mollusks were observed in the bare site of *Sonneratia alba* J. Smith with the reforested site and natural reference having similar species composition, density and diversity. The lack of mollusks in the bare site emphasizes the consequences of mangrove degradation on biodiversity, while similarities among the replanted site and natural reference suggest the potential of mangrove restoration in enhancing faunal recolonisation.

Studies of epibiotic communities in Kenya (Crona et al., 2006) compared natural stands with two 8-year-old *Sonneratia alba* plantations; an integrated plantation (a reforested stand originally degraded site but with some remaining mangroves) and a matrix plantation (a reforested stand which was originally clear-felled). The study showed a decreasing trend of similarity with natural stands when comparing macroalgal assemblages of an integrated plantation, a matrix plantation and a clear felled area, in this order.

Both algal diversity and root fouling faunal cover and biomass were lower in the matrix plantation compared to the integrated plantation and natural stand which was attributed to lower root area, in combination with longer inundation times and larval behaviour and longevity of poriferans and barnacles, which may affect recruitment patterns.

3.2. Sediment-infauna

Sediment-infauna communities showed patterns similar to those described above. Lower diversity of taxa was observed in planted versus natural sites in Qatar with the picture being less clear in Thailand. Infaunal studies in the Matang mangroves in Malaysia suggested that 2-year-old planted mangroves had the greatest biomass and species number followed by the mature and 15-year-old stand, although species diversity was highest in the mature site and lowest in the 2-year-old site (Sasekumar and Chong, 1998). In Kenya, bare sites of *A. marina*, *R. mucronata* and *S. alba* had the lowest infauna densities and taxa richness compared to respective replanted sites with conspecific natural references having the highest densities. Taxa richness and composition were similar among respective replanted and natural sites (Bosire et al., 2004), suggesting successful fauna recolonisation following mangrove restoration.

3.3. Vagile fauna–fish and shrimp

Mangroves support nursery functions for many juvenile fish and shrimp species, many of which are highly important commercially (Lewis et al., 1985; Rönnbäck et al., 1999; Nagelkerken et al., 2008). Juvenile fish and shrimp species are known to be dependent on structural complexity for refuge (Primavera, 1997) and therefore the intensity of this function is linked to the type of mangrove in focus (Ewel et al., 1998; Rönnbäck et al., 2001). Studies of vagile fauna in replanted mangroves of varying ages and species composition showed variable patterns. In Qatar, lower diversity of both juvenile and adult fish was observed in plantations compared to natural stands of *A. marina* (Al-Khayat and Jones, 1999). Studies comparing fish and shrimp density between natural stands of *R. apiculata* Bl., *Avicennia officinalis* L., *A. marina* and a single replanted *R. apiculata* stand (5–6 years old) in the Philippines indicated that density and biomass were primarily influenced by tidal height and mangrove species (Rönnbäck et al., 1999). In *S. alba* plantations in Kenya, there were strong seasonal fluctuations for juvenile fish, showing temporal patterns to be a potentially stronger influence on fish assemblages than type of plantation or presence of fringing mangroves (Crona and Rönnbäck, 2007). However, the spatial scale of observation is likely a much stronger factor affecting biodiversity studies of plantations for vagile fauna compared to less mobile communities described above. Since most studied plantations are small in size, the effect of plantations on fish distribution patterns therefore remains largely unknown. The same is true for juvenile shrimps. In Kenya lower species richness was observed in a matrix plantation of *S. alba*, and in adjacent clear felled areas one species, *Penaeus japonicus* (Bate), dominated the community (Crona and Rönnbäck, 2005). Natural forests had higher root complexity and also higher abundances and more even distribution of shrimp species in terms of species composition. Similarly, in the Philippines, higher abundances of juvenile shrimp in a planted *R. apiculata* site were seemingly related to higher structural root complexity, although more inland stands of mature *Avicennia* spp. and *Rhizophora* spp. showed no such differences and had equally high densities as the near-shore *Rhizophora* spp. (Rönnbäck et al., 1999).

Few studies exist on trends in biodiversity in restored mangroves, and the range in age, species and inundation class of restored sites makes generalizations hard. However, the co-occurrence of many animal species in both restored and comparable natural forests suggest recovery of the former sites. Lewis (1992) in reviewing both tidal marsh and mangrove restoration projects in the United States noted that the recovery of fish populations back to similar species composition and density as reference sites has been accomplished within 5 years. To optimize fish habitat in mangrove restoration projects, Lewis and Gilmore (2007) have recommended restoration of tidal creeks to provide access and low tide refuge for mobile nekton.

Although the results reviewed in this section are quite variable most likely due sampling design and intensity, in most cases they suggest remarkable recovery of biodiversity in restored mangroves. It is also apparent that mangrove degradation causes not only a general decline in species richness and/or diversity, but also functional shifts as sets of species with particular traits are replaced. Some higher order groups have also been found to be more sensitive to mangrove degradation, e.g. sesarmid crabs and mollusks. This suggests that while abundance and diversity are important measures of biodiversity, species composition as an analogue to functional diversity, may be an additional, more objective and distinct index of measuring faunal recovery in restored mangroves. To make data obtained from various locations comparable, it will be necessary for teams involved in mangrove restoration ecology to agree on standard approaches to measure recovery of biodiversity. Currently these do not exist.

4. Socio-economics of mangrove restoration

The socio-economic importance of natural mangrove goods and services has been documented repeatedly (Ruitenbeek, 1994; Walters, 1997; Adger et al., 2001; Barbier, 2006; Walters et al., 2008), but can restored mangroves generate income similar to that of natural mangroves? To date there have been insufficient studies in replanted mangroves to be sure and comparisons are further complicated by the diversity in productivity of natural mangrove habitats.

Mangroves were initially planted in order to generate income from timber. At Matang in Malaysia one of the best-managed mangrove plantations can be found (Gong and Ong, 1995). Here, 17.4 t ha^{-1} $year^{-1}$ of mangrove wood is harvested sustainably over a 30-year cycle (Gan, 1995). A similar study in Java suggested that a 7-year-old *R. mucronata* plantation had a standing trunk and branch biomass of 74 t ha^{-1}, and a production of 10.6 t ha^{-1} $year^{-1}$ (Sukardjo and Yamada, 1992).

Governments are increasingly aware of the nursery and fisheries enhancement function of mangroves. In the Mekong Delta, Soc Trang province, Vietnam, extensive planting of *Rhizophora* species was used as a coastal protection measure. Recent studies here in a 7 ha area reforested in 1995 with *R. apiculata* suggested an annual harvest rate of fish and crustaceans of 143 kg^{-1} ha^{-1} $year^{-1}$ valued at USD 363 ha^{-1} $year^{-1}$ (Walton and Le Vay, unpublished, 2006).

Recently a questionnaire-based socio-economic study on the Buswang replanted mangroves in the Philippines suggested the mangrove was directly benefiting local incomes in the region of USD 564–2316 ha^{-1} $year^{-1}$ (Walton et al., 2006a). Contributing to the annual income are mollusc, crustacean and fish catches from within the mangroves (294 kg ha^{-1} worth USD 213 ha^{-1}), tourism (USD 41 ha^{-1}), timber (USD 60 ha^{-1}) and an estimate of contribution of these mangroves to near-shore coastal mangrove associated fish (10% to 276 kg ha^{-1} worth USD 250 ha^{-1} to 80% to 2204 kg ha^{-1} worth USD 2002 ha^{-1}). The increase in interest in carbon credits could in the future also raise an additional income of USD 163–198 ha^{-1} $year^{-1}$ (Walton et al., 2006a). These fisheries values compare favourably with those from natural mangroves that are estimated to support fisheries valued at between USD 750–11,280 ha^{-1} year^{-1} (Rönnbäck, 1999).

Other indirect benefits such as coastal protection and non-use values (option, bequest and existence values) are more difficult to gauge. Since the establishment of the Buswang mangrove, storm surge damage and coastal erosion has been negligible, but in some other countries around the Indian Ocean, cases about storm-associated costs have been documented (cf. Gilman et al., 2008). In India for instance, monetary losses due to repair and reconstruction costs of personal property (incl. livestock and agricultural products) ranged between 32 USD per household in mangrove-protected villages to 154 USD per household in villages that were not protected by mangroves (Badola and Hussain, 2005). In the past, replacement costs have been used to estimate coastal protection value. However replacement cost associated with constructed breakwaters generally overestimate the value. As such this should be modified by the area that requires coastal protection estimated as USD 3679 ha^{-1} year^{-1} (Sathirathai and Barbier, 2001). Other indirect benefits include accretion of agricultural land. In the Sundarbans, Bangladesh, the planting of 150,000 ha of mixed mangrove species has enhanced the deposition of sediments to such an extent that the elevation of 60,000 ha is no longer suitable for mangrove, and can be used for agriculture worth US$ 800 ha^{-1} year^{-1} (Saenger and Siddiqi, 1993). However, it remains to be seen to which extent novel functions gained, such as from agriculture, outweigh their possibly adverse ecological impacts on the mangrove.

While the total extent of the economic benefit of restored mangroves is as yet unclear, the initial planting costs are a major factor in preventing more community based replanting efforts. In the Philippines, initial costs are estimated to be USD 204 ha^{-1} using volunteers (Walton et al., 2006a). However mangrove restoration cost estimates for the United States of America ranged between 225 and 216,000 USD ha^{-1} (Lewis, 2005). These costs thus vary very widely depending on differential labour costs (dependent on GNP of the country in question (Brander et al., 2006), site conditions and thus the effort in terms of labour required for hydrological restoration and removal of debris and weeds among other factors, and planting material types where replanting is necessary. Grant-based aid and elimination of ownership doubts through community stewardship schemes could significantly boost mangrove replanting programs around the world.

5. Opportunities and constraints to mangrove forest restoration

Mangrove forest ecosystems currently cover an estimated 14.7 million ha of the tropical shorelines of the world (Wilkie and Fortuna, 2003). This represents a decline from 19.8 million ha in 1980 and 15.9 million ha in 1990. These losses represent about 2% year^{-1} between 1980 and 1990, and 1% year^{-1} between 1990 and 2000. Therefore achieving no-net-loss of mangroves worldwide would require the successful restoration of approximately 150,000 ha year^{-1}, unless all major losses of mangroves ceased. Increasing the total area of mangroves worldwide would require an even larger scale effort.

Recently, Duke et al. (2007) sounded once more the alarm bell and indicated that a world without mangroves is a realistic forecast if the destruction of mangrove ecosystems continues. Examples of documented losses include combined losses in the Philippines, Thailand, Vietnam and Malaysia of 7.4 million ha of mangroves (Spalding et al., 1997). These figures emphasize the level of opportunities that exist to restore larger areas of mangroves such as mosquito control impoundments in Florida (Brockmeyer et al., 1997) (tens of thousands of ha), and abandoned shrimp aquaculture ponds in Southeast Asia (Stevenson et al., 1999; hundreds of thousands of ha), back to functional mangrove ecosystems.

However while great potential exists to reverse the loss of mangrove forests worldwide, most attempts to restore mangroves often fail completely, or fail to achieve the stated goals (Elster, 2000; Erftemeijer and Lewis, 1999; Lewis, 2000, 2005).

Restoration or rehabilitation may be recommended when an ecosystem has been altered to such an extent that it can no longer self-correct or self-renew. Under such conditions, processes of secondary succession or natural recovery are inhibited in some way. Secondary succession depends upon mangrove propagule availability. Lewis (2005) proposed a new term, "propagule limitation" to describe situations in which mangrove propagules may be limited in natural availability due to removal of mangroves by development, or hydrologic restrictions or blockages (i.e. dikes) which prevent natural waterborne transport of mangrove propagules to a restoration site. In Sri Lanka, such hydrographical alterations have resulted in a decrease in forest flooding frequency by >90% (Dahdouh-Guebas, 2001). Predation on natural propagules can also limit their availability and indicate that broadcasting of collected seeds or planting may be essential for successful restoration (Dahdouh-Guebas et al., 1997, 1998; Bosire et al., 2005b; Cannicci et al., 2008).

Restoration has, unfortunately, emphasized planting mangroves as the primary tool in restoration, rather than first assessing the causes for the loss of mangroves in an area, then assessing the natural recovery opportunities, and how to facilitate such efforts. Thus most mangrove restoration projects move immediately into planting of mangroves without determining why natural recovery has not occurred. There may even be a large capital investment in growing mangrove seedlings in a nursery before existing stress factors at a proposed restoration site are assessed. This often results in major failures of planting efforts (Elster, 2000; Erftemeijer and Lewis, 1999; Lewis, 2005). In addition, few restoration efforts are embedded in a larger framework that also considers the fate of the planted mangroves, in terms of stand structure and regeneration, return of biodiversity and recovery of other ecosystem processes (Dahdouh-Guebas and Koedam, 2002). Recently these questions are starting to be tackled in an integrated way in East-African restored mangrove sites (Bosire et al., 2003, 2004, 2005a,b; Crona and Rönnbäck, 2005; Bosire et al., 2006; Crona et al., 2006).

Although a number of papers discuss mangrove hydrology (Kjerfve, 1990; Wolanski et al., 1992; Furukawa et al., 1997), their focus has been on tidal and freshwater flows within the forests, and not the critical periods of inundation and dryness that govern the health of the forest. Kjerfve (1990) does discuss the importance of topography and argues that "...micro-topography controls the distribution of mangroves, and physical processes play a dominant role in the formation and functional maintenance of mangrove ecosystems..." The point of all of this is that flooding depth, duration and frequency are critical factors in the survival of both mangrove seedlings and mature trees (Thampanya et al., 2003; Bosire et al., 2006), and also determine many of the functional attributes, like crustacean and fish use of forests. Once established, mangroves can be further stressed if the tidal or freshwater hydrology is changed, for example by diking (Brockmeyer et al., 1997; Dahdouh-Guebas et al., 2000a,b, 2005). Both increased salinity due to reductions in freshwater availability, and flooding stress, increased hypoxic or anoxic conditions and free sulfide availability can kill existing stands of mangroves. However, also increases in freshwater availability may result in a shift in species composition which favours ecologically and economically inferior species (Dahdouh-Guebas et al., 2005). The consulted scientist should therefore pay attention to both ecological and socio-economic functions of the mangrove stand or the restoration site in question.

Ecological restoration of mangrove forests has only received attention very recently (Lewis, 2000). The wide range of project types previously considered to be restoration, as outlined in Field (1996, 1998), reflect the many aims of classic mangrove rehabilitation or management for direct natural resource production. As mentioned previously, these include planting monospecific stands of mangroves for future harvest as wood products. This is not ecological restoration as defined by Lewis (2005).

Because mangrove forests may recover without active restoration efforts, it has been recommended that restoration planning should first look at the potential existence of stresses such as blocked tidal inundation that might prevent secondary succession from occurring, and plan on removing that stress before attempting restoration (Hamilton and Snedaker, 1984; Cintron-Molero, 1992). The next step is to determine whether natural seedling recruitment is occurring once the stress has been removed. Assisted natural recovery through planting should only be considered if natural recovery is not occurring.

Lewis and Marshall (1997) first suggested six critical steps necessary to achieve ecological mangrove restoration, and these are discussed in more detail in Stevenson et al. (1999). The general approach is to emphasize careful examination of factors hindering natural regeneration restoration opportunities while avoiding emphasizing planting of mangroves (Turner and Lewis, 1997). These steps have been tested in training courses on mangrove restoration in the USA and India, and have been further modified to support site-specific ecological restoration. However, the steps above have hitherto ignored the human dimension as an important consideration in mangrove restoration projects. In this paper we therefore further develop these steps into a functional framework which incorporates the human dimension (Fig. 2).

Mangrove forests may recover without active restoration efforts. When natural regeneration fails and the process needs human intervention, an understanding of the autoecology and community ecology of the targeted mangrove species is necessary, i.e. its reproductive patterns, propagule dispersal, seedling establishment, zonation and hydrology (steps 1 and 2). With this understanding, an assessment of factors hampering successful secondary succession can be done (step 3), involving the local knowledge of communities depending on the mangroves (step 4), which will be relevant throughout the subsequent steps. The perceptions and expectations of the local community depending on the mangroves should be considered during mangrove planting (cf. Dahdouh-Guebas et al., 2006). Coastal populations in industrialized countries typically do not depend on mangroves for their daily livelihoods, but in the majority of mangrove countries (developing countries) they do. The concerns of the local people in terms of how dependent they are on the mangroves, which species preferences do they have, and which alternatives can be offered while the natural ecosystem is left to recover or a planted site is left to develop can be captured through socio-ecological surveys, which can then be integrated in the restoration exercise (Dahdouh-Guebas, 2008). The surveys can also yield fundamental social and economic drivers of deforestation, which are equally important to restoration as hydrology. More specifically, the perceived value among local users of the ecosystem goods and services provided by mangroves to their overall livelihoods is essential if socio-economic drivers of degradation are to be altered or decreased (Rönnbäck et al., 2007).

The socio-ecological information gathered from steps 3 and 4 is then used to select appropriate restoration sites (step 5), and the obstacles to successful natural regeneration removed (step 6). If conditions are favourable, this should allow natural revegetation (successful aided natural regeneration) of the site, which is more cost-effective than replanting. If natural revegetation fails despite all these interventions (cf. Dahdouh-Guebas and Koedam, 2008,

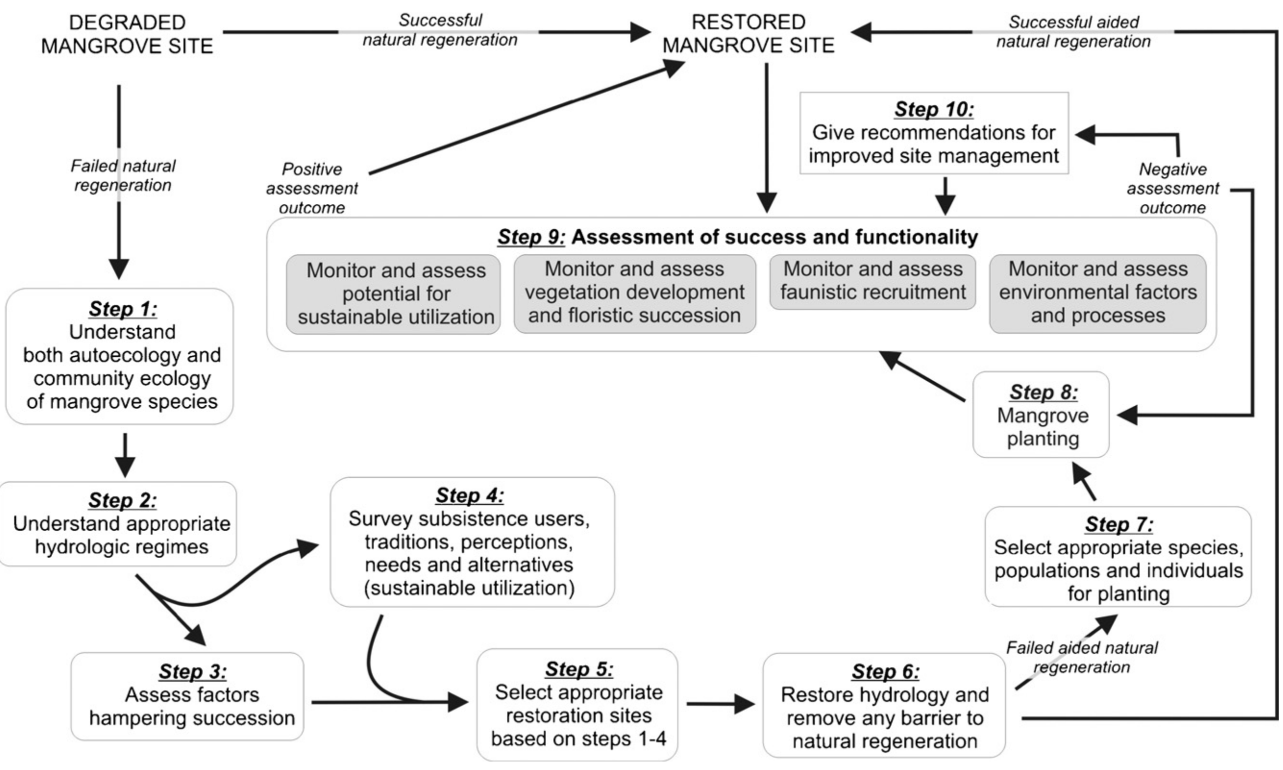

Fig. 2. A 10 steps scheme presenting possible mangrove restoration pathways depending on site conditions (modified after Stevenson et al., 1999; Bosire et al., 2006).

Fig. 1), then appropriate mangrove species, populations and individuals for planting (step 7) must be selected in view of genetic diversity (Triest, 2008), faunistic impacts (Cannicci et al., 2008) and individual performance (Komiyama et al., 2008) and mangrove replanting (step 8) becomes necessary to restore the degraded site. At regular intervals the replanting effort should be assessed for four different key factors in mangrove ecosystem functioning (step 9): the flora, the fauna, the environment and human subsistence use. When the assessment has a negative outcome, recommendations should be given for improved site management (step 10), which may have to be accompanied by extra planting. When the assessment has a positive outcome the site has restored, although further monitoring of the restored site can be undertaken as necessary.

The assessment of success of restoration is an essential step that is unfortunately backed up by few scientific papers (Walters, 2000; Macintosh et al., 2002; Bosire et al., 2004, 2006; Crona and Rönnbäck, 2005; Crona et al., 2006; Walton et al., 2006a,b; Lewis and Gilmore, 2007). We recommend that four assessment types are necessary as indicators of restoration success: development of the vegetation and floristic succession, faunistic recruitment, evolution of environmental factors and processes, and finally the potential for sustainable exploitation. The first three indicators can be started soon after the initiation of the natural recovery or plantation and can be repeated regularly (Bosire et al., 2006), whereas the last one is on a longer term of >10 or even >20 years. All of these can be assessed by using natural sites (references) under the same conditions on one hand, and to bare sites lacking mangroves on the other hand, as discussed in the preceding sections.

From the foregoing, it is clear that the two primary factors in designing a successful mangrove restoration project are habitat conditions (e.g. hydrology, herbivory and weed cover among others) as well as the participation of local communities from the onset of the restoration initiative. Community involvement is likely to increase the legitimacy of the restoration project and increase the likelihood of future sustainable use and compliance with regulatory measures to protect the developing stands of restored mangroves (Rönnbäck et al., 2007). Determination of appropriate hydrology (depth, duration and frequency of tidal flooding) of existing natural mangrove plant communities (a reference site) in the area in which you wish to do restoration is a critical factor. For instance, Vivian-Smith (2001) recommends the use of a reference tidal marsh for restoration planning and design. A common surrogate for costly tidal data gathering or modeling is the use of a tidal benchmark and survey of existing healthy mangroves. Similar topography is then established at the proposed restoration site, normal hydrology restored to a diked site, or tidal streams reestablished or created at damaged sites to ensure proper drainage, propagule dispersal and faunal access during tidal flooding.

A question that needs to be addressed in contemporary mangrove restoration projects is whether monospecific planting is appropriate in all situations. Considering that mixed species stands, even if dominated by few species, are common (e.g. mosaic mangrove stands in Sri Lanka, Dahdouh-Guebas et al., 2000a,b), one should consider the possibility that in some reforestation projects, monospecific planting may not be ideal, and even counter-advised. In the Philippines, an extensive area of monospecifically replanted *Rhizophora* spp. was lost due to an attack by tussock moth larvae (Walton et al., 2006b). Modelling vegetation development and individual interactions may be a helpful tool in the entire restoration framework (cf. Berger et al., 2008).

In summary, maintaining a no-net-loss of mangrove habitat worldwide will require very large scale restoration efforts which demand a common ecological engineering approach and application of the steps to successful restoration outlined above. This would ensure an analytically thought process and less use of small scale "gardening" of mangroves as the solution to all mangrove restoration problems. Those involved could then begin to learn more from past successes or failures, act more effectively based on this knowledge, and spend limited mangrove restoration funds in a more cost-effective manner. It will also be important to define criteria for monitoring mangrove restoration projects to include main ecosystem attributes namely: biodiversity, vegetation structure and ecological processes (Ruiz-Jaen and Aide, 2005).

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