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Review

Ethnobiology, socio-economics and management of mangrove forests: A review

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ABSTRACT

There is growing research interest in the ethnobiology, socio-economics and management of mangrove forests. Coastal residents who use mangroves and their resources may have considerable botanical and ecological knowledgeable about these forests. A wide variety of forest products are harvested in mangroves, especially wood for fuel and construction, tannins and medicines. Although there are exceptions, mangrove forest products are typically harvested in a small-scale and selective manner, with harvesting efforts and impacts concentrated in stands that are closer to settlements and easiest to access (by land or by sea). Mangroves support diverse, local fisheries, and also provide critical nursery habitat and marine productivity which support wider commercial fisheries. These forests also provide valuable ecosystem services that benefit coastal communities, including coastal land stabilization and storm protection. The overlapping of marine and terrestrial resources in mangroves creates tenure ambiguities that complicate management and may induce conflict between competing interests. Mangroves have been cut and cleared extensively to make way for brackish water aquaculture and infrastructure development. More attention is now given to managing remaining forests sustainably and to restoring those degraded from past use. Recent advances in remotely sensed, geo-spatial monitoring provide opportunities for researchers and planners to better understand and improve the management of these unique forested wetlands.

Contents

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1. Introduction

Mangroves have been extensively studied for decades by botanists, ecologists and marine scientists (Macnae, 1968; Chapman, 1976; Saenger et al., 1983; Tomlinson, 1986; Kathiresan and Bingham, 2001; Lacerda, 2002). Yet, it was not until the 1980s and early 1990s that significant research attention was brought to bear on the human interactions with these unique forested wetlands (FAO, 1985; Hamilton et al., 1989; FAO, 1994; Cormier-Salem, 1999). Earlier works were mostly descriptive, documenting the status and uses of mangroves by coastal communities (e.g., Walsh, 1977; Taylor, 1982; Christensen, 1982; Kunstadter et al., 1986; Field and Dartnall, 1987; Diop, 1993; Lacerda, 1993). By contrast, recent research on mangroves is more analytical, examining humans as ecological agents of disturbance and change in mangrove ecosystems. These studies have applied a mix of ecological, economic, ethnographic, historical and geo-spatial methods to quantify the diverse values of mangrove forests and to probe cause–effect relationships between people and mangroves in a variety of geographic, cultural and political-economic contexts (e.g., Dewalt et al., 1996; Ellison and Farnsworth, 1996; Ewel et al., 1998b; Rönnbäck, 1999; Vandergeest et al., 1999; Kovacs, 2000; Barnes, 2001; Walters, 2003, 2005b; Dahdouh-Guebas et al., 2006a; Lopez-Hoffman et al., 2006).

This review paper synthesizes research on the ethnobiology, socio-economics and management of mangrove forests, and also includes a brief review of geo-spatial monitoring tools as these have been applied to study mangroves. These topics span an enormously diverse range of literature. As such, different subtopics are necessarily dealt with succinctly. An attempt was made to include the most significant publications as well as a good number of the less noted, but also important research works. The extensive bibliography can serve as a resource for readers interested in further exploration of the subject.

Population pressure is typically greatest along the coast, so it is little surprise that human influences on the world's mangrove forests are significant and growing. Mangroves have been cleared and degraded on an alarming scale during the past four decades (Valiela et al., 2001; Wilkie and Fortuna, 2003; Duke et al., 2007), yet they remain an important source of wood and food products and provide vitally important environmental services for coastal communities throughout the tropics (Balmford et al., 2002). These values still receive relatively little attention or recognition from government policy-makers and the development community, and the myriad influences people have on these forests continue to be overlooked by many mangrove researchers. It is hoped that this review paper will provide some corrective to this neglect.

2. Ethnobiology of mangroves

Local ecological knowledge (LEK) or traditional ecological knowledge (TEK) are closely related concepts that are broadly inclusive of many different types of ecologically relevant knowledge, ranging from traditional use of specific plants and animals and essential knowledge critical to harvesting natural resources, through complex understandings of the functioning of local ecosystems, to cultural beliefs and religious views of human–environment relations (Berkes, 1999; Davis and Wagner, 2003).

There is an implicit assumption that most LEK is accumulated through experiences of close contact with the natural environment, and therefore locality plays a large part in shaping this knowledge (Davis and Wagner, 2003). The local scale has also been shown to be important in resource extraction patterns and resulting impacts on mangroves (Tomlinson, 1986; Ewel et al., 1998b; Kovacs, 1999; Dahdouh-Guebas et al., 2000a, 2000b, 2006a; Walters, 2005a, 2005b; Lopez-Hoffman et al., 2006). The role of LEK in shaping resource use in mangroves is therefore of great interest for management of these ecosystems. There is much opportunity to integrate indigenous knowledge into contemporary frameworks for conservation and sustainable management, or in a *priori* understanding of forest dynamics and local dependency using ethnoscientific approaches (Rist and Dahdouh-Guebas, 2006) and modeling (Berger et al., 2008). Studies of mangrove LEK and ethnobiology can be split into two general categories: one focusing on the functioning of the ecosystem, including knowledge of ecological processes and how different ecological components interact with each other; the other focusing more on specific species or taxa and their use for anthropocentric purposes, often termed ethnotaxonomy or ethnobotany (Berlin, 1973).

Studies in Mexico, the Philippines, Tanzania, Kenya, India and Venezuela are worth briefly describing as examples where LEK representing basic ecosystem dynamics has been documented. Kovacs (2000) showed how Mexican fishermen have extensive knowledge of mangrove system dynamics, including previously undocumented sources of local environmental disturbance that help explain changes in the forest over time. Similarly, Walters (2003, 2005b) sought the knowledge of local fishermen and coastal residents in the Philippines to assist in mapping and explaining changes to the distribution of mangrove forests. Tobisson et al. (1998) found intricate LEK within Zanzibar fishing communities relating to tidal patterns and currents, but linked to mangroves and associated fisheries. In Kenya, Crona (2006) similarly showed a large body of LEK related to complex ecological linkages between mangroves and the surrounding seascape, and noted marked differences in local peoples' knowledge based on their gear types and modes of resource extraction from the mangrove. This heterogeneous distribution of LEK between user groups is a common theme throughout much LEK work on mangroves and other systems (Kovacs, 2000; Dahdouh-Guebas et al., 2000b; Ghimire et al., 2004; Vayda et al., 2004; Walters, 2004; Hernández Cornejo et al., 2005; Dahdouh-Guebas et al., 2006a). The benefit of such heterogeneity and spatially distributed LEK is that it can be valuable for documenting and understanding variations in patterns of mangrove use and change that would otherwise not be apparent with larger-scale scientific assessments and monitoring (Kovacs, 2000).

Understanding of ecosystem dynamics by local communities has also proven valuable as a background to reconstruct historical use and impact on mangroves (Walters, 2003; Dahdouh-Guebas et al., 2004, 2005b), although efforts should be made to validate such information before it is applied to policy and management decisions (Kovacs, 2000; Hernández Cornejo et al., 2005). Validation, in this sense, means sound interpretation by cross-checking statements with other information sources, including pre-existing historical documents, data from remotely sensed imagery and modeling, and experimental field-testing (Kovacs et al., 2001a, b; Vayda et al., 2004; Hernández Cornejo et al., 2005; Bart, 2006; Lopez-Hoffman et al., 2006). This historical aspect of LEK can, when used in conjunction with scientific results, also increase the chance of including important ecological information potentially missed by short-term duration scientific studies (Moller and Berkes, 2004; Bart, 2006). Examples of this can be seen in findings on the role of caterpillars and hurricanes as agents of mangrove forest disturbance in Mexico (Kovacs, 2000), and in information on sea urchin infestations in Kenya (Crona, 2006).

The second knowledge category is represented by ethnobotany which relates to taxonomy and use of specific plants for different purposes. This is a better-documented field than the LEK of system dynamics reviewed above, although very fragmentary from a global perspective. In many coastal communities, mangrove dependence is high and both wood and non-wood products are used for a multitude of purposes. Discussions of LEK as this pertains to mangrove resource use are embedded in subsequent sections of the paper that detail forest and aquatic resource uses. Nonetheless, a few general comments and examples are warranted here.

Like the aforementioned studies on knowledge of basic ecology, LEK that is related to mangrove resource use is often well developed, but heterogeneous between and within coastal communities in ways that typically reflect their varied experience and dependence on the use of particular resources. For example, Lopez-Hoffman et al. (2006) found sharp differences in the perceptions and practices of older, more experienced versus younger, less experienced mangrove wood harvesters in Venezuela. The same is true for Kenyan mangrove users, as those with greater experience were better able than others to identify forest vegetation decline (Dahdouh-Guebas et al., 2000b). Similarly, studies of coastal residents in the Philippines who were engaged in the local silviculture of mangrove trees revealed that knowledge among planters about propagation and management was considerable, but varied enormously depending on personal experience and opportunities to learn from others more knowledgeable. The differences in knowledge had significant consequences for the relative success of individual mangrove tree planters (Vayda et al., 2004; Walters, 2004).

However, as knowledgeable as local people were sometimes found to be, it is notable that mangrove users in the aforementioned Venezuelan and Philippine cases were sometimes found to act in ways that were inconsistent with their knowledge and avowed beliefs by, for example, over-cutting and clearing mangroves that they otherwise viewed as important to protect (Vayda and Walters, 1999; Walters, 2004; Lopez-Hoffman et al., 2006). This gap between knowledge and behavior, also known as 'cognitive dissonance' (Festinger, 1957), is displayed by most humans to various degrees and is often caused by conflicting interests or incentives. While this does not invalidate the LEK per se, such knowledge should not be assumed to always guide the behavior of local users in terms of resource use, etc. (Vayda et al., 2004; Bart, 2006). Economic incentives, property rights and participation in the management process are also likely to influence such behavior.

3. Mangrove forest products: use and consequences

3.1. Mangrove forest users and uses

Non-timber forest products are recognized as important economic resources, particularly to rural, marginalized communities (Vedeld et al., 2004). Many coastal communities in the tropics are characterized by relative geographic isolation, chronic poverty and significant dependence on the harvest of marine and coastal resources for their livelihood (Kunstadter et al., 1986). The majority of people living in or near mangrove areas derive their principal income from fishing and related activities. The direct harvest of mangrove wood and plants is rarely a full-time occupation for them, but a great many rely on these products to meet subsistence needs for fuel and construction materials, and for others the harvest and sale of mangrove forest products is an important income supplement (Christensen, 1982; FAO, 1985, 1994; Kunstadter et al., 1986; Diop, 1993; Lacerda et al., 1993; Spalding et al., 1997; Glaser, 2003; Walters, 2005a; Lopez-Hoffman et al., 2006; Rönnbäck et al., 2007a).

The two most widespread uses of mangrove wood are for fuel and construction. Many common mangrove tree species, e.g., *Rhizophora* species produce wood that is dense, hard and often rich in tannins (FAO, 1994; Bandaranayake, 1998). Such wood burns long and hot, and so is highly attractive for making charcoal or consuming directly as firewood (Brown and Fischer, 1918; Chapman, 1976; Christensen, 1982, 1983b; Taylor, 1982; Bhattacharyya, 1990; Ewel et al., 1998a; Walters, 2005a; Dahdouh-Guebas et al., 2006a). The harvest of mangrove for fuelwood is widespread throughout the coastal tropics (Fig. 1A and D). In some countries, mangrove wood historically formed an important commercial fuel for industries like bakeries and clay-firing kilns, although this is less common today because of the ready availability of alternative fuels, like natural gas and electricity, and policies aimed at discouraging mangrove cutting (Lacerda et al., 1993; Naylor et al., 2002; Walters, 2003). Nonetheless, remote coastal communities in many parts of the tropics continue to depend heavily on mangrove wood for domestic fuelwood consumption, and commercial markets that sell mangrove charcoal to nearby towns and urban centers are not uncommon (Untawale, 1987; Walters and Burt, 1991; Alvarez-Leon, 1993; Allen et al., 2000; Dahdouh-Guebas et al., 2000b; Glaser, 2003).

The qualities of strength and durability (including pest- and rot-resistance) also make mangrove wood well-suited for use in construction (Adegbehin, 1993; Bandaranayake, 1998; Kairo et al., 2002; Walters, 2005a). Yet, the typically short and contorted growth form of tree stems of common genera such as *Avicennia* and *Sonneratia* renders them of limited value for large, commercial-sized lumber. The extraction of construction wood from mangroves is thus limited mostly to domestic consumption and sale of small-size posts to targeted local and regional markets (Fig. 1C). Mangrove wood is widely used in coastal communities for residential construction (posts, beams, roofing, fencing) and to make fish traps/weirs (Adegbehin, 1993; Alvarez-Leon, 1993; Rasolofo, 1997; Ewel et al., 1998a; Semesi, 1998; Kovacs, 1999; Primavera et al., 2004; Walters, 2004). Fronds from the mangrove "nipa" palm (*Nypa fruticans* (Thunb.) Wurmb.) are particularly valued in Southeast Asia for use in roofing and as thatch in walls and floor mats (Aksornkoae et al., 1986; Fong, 1992; Basit, 1995; Spalding et al., 1997; Walters, 2005a). Mangrove wood is also used in some countries for building boats, furniture, wharf pilings, telegraph poles, construction scaffolding, railway girders and mine timbers (Walsh, 1977; Mainoya et al., 1986; Adegbehin, 1993; Bandaranayake, 1998; Primavera et al., 2004; Lopez-Hoffman et al., 2006).

In addition to wood for fuel and construction, mangrove forest trees are also widely valued for their bark (used in tanning and dyes)

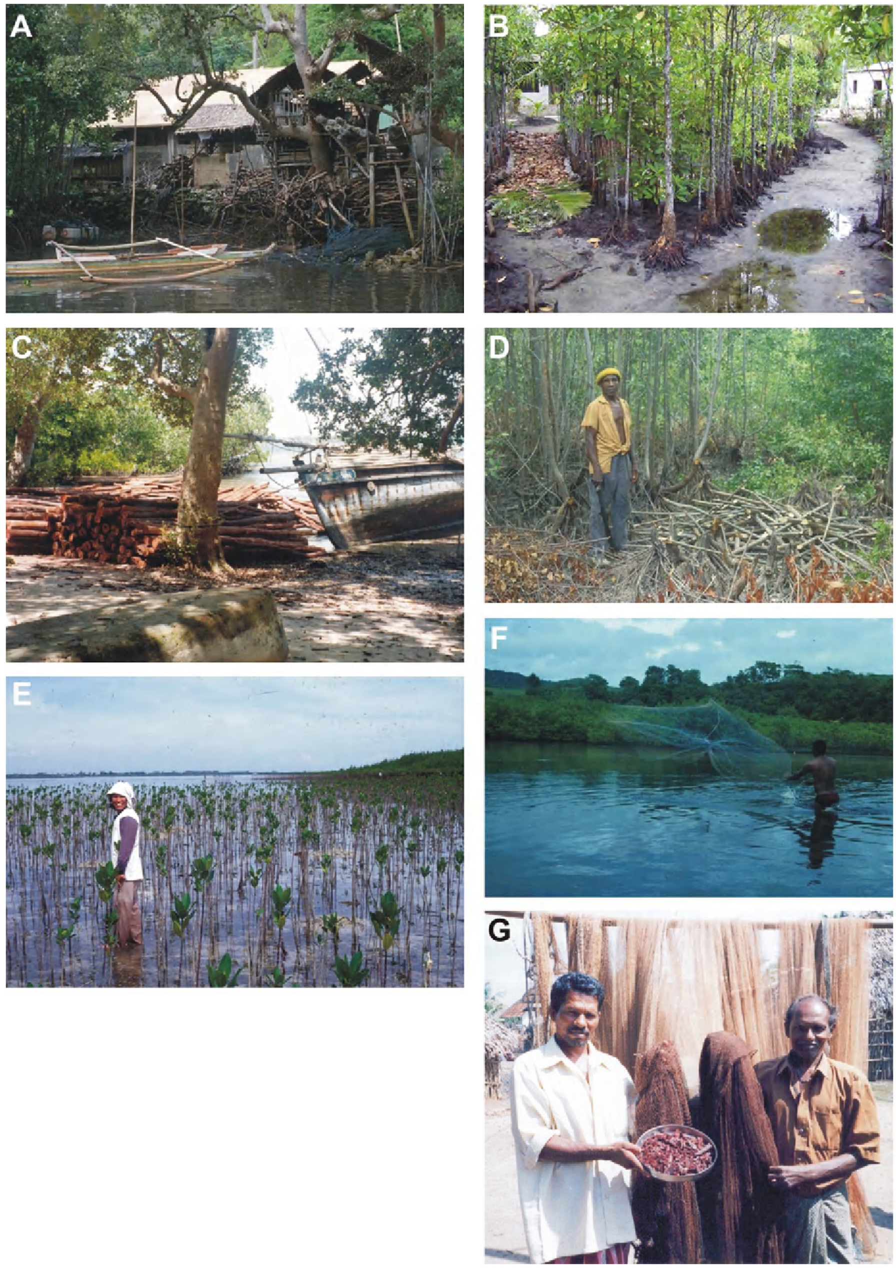

Fig. 1. (A) Fishermen in Bais Bay, Philippines commonly build their homes adjacent to mangroves where they gain ready access to wood products and favored fishing spots, and benefit from the storm protective value of mangrove trees. (B) An illustration of the concept of living in mangroves in Balapitiya, Sri Lanka: houses were built within a mangrove and ***Bruguiera gymnorrhiza*** assemblages were cut in such a way that they form access paths to each house. (C) Mangrove poles at the Sita landing place in Mida Creek, Kenya waiting to be transported to markets and hardware stores. (D) Mangroves in Mankote, Saint Lucia are often cut to make charcoal, a fuel preferred by many West Indians for barbecuing. (E) Gleaners like this woman on Banacon Island, Philippines are free to harvest for shellfish within a plantation of ***Rhizophora stylosa*** as long as they do not disturb the young trees. (F) Simple fishing techniques like this throw-net are effective for capturing fish in the murky, brackish waters of the Mankote mangrove, Saint Lucia. (G) Fishermen holding a tray with pieces of ***Ceriops decandra*** bark used for dyeing fishing nets near Kakinada in Andhra Pradesh, India. They also show two freshly dyed nets and in the background previously dyed nets are hung to dry. Adopted from Dahdouh-Guebas (2006). (***Note***: photos in Fig. 1A and D–F by Brad Walters; (B), (C) and (G) by Farid Dahdouh-Guebas).

and wood fiber (to make rayon and paper); as sources of animal fodder, vegetable foods, and diverse traditional medicines and toxicants (see Bandaranayake, 1998, 2002 for a reviews); and as habitats for honey bees and hunted wildlife (see Table 1; Fig. 1G).

3.2. Patterns and consequences of forest use

Different mangrove species have different wood properties, making some more suitable than others for specific uses (FAO, 1994). For example, trees from the Rhizophoraceae family (*Rhizophora*, *Ceriops*, *Bruguiera*) are characterized by hard, dense wood that is rich in tannins and, as such, is widely valued for construction, fuelwood and tannin extraction, yet this wood is not suitable for lumber or furniture-making because of its tendency to split (Ewel et al., 1998a). Studies have documented mangrove wood harvesting that is size- and species-selective, and harvesters willing to venture widely in search of particular trees that are used in construction and have high local market value (Rasolofo, 1997; Dahdouh-Guebas et al., 2000b; Hauff et al., 2006).

However, despite differences in wood character and quality, research suggests that mangrove wood users are often flexible in their preferences, and willing to substitute favored mangrove

Table 1
Summary of mangrove forest products and uses, with selected published references

Forest products and use	Selected references
Wood for fuel (charcoal, firewood)	See text
Wood for construction materials	See text
Tree bark for tannins, dyes	Chapman, 1976; Aksornkoae et al., 1986; Mainoya et al., 1986; Lacerda et al., 1993; Dahdouh-Guebas et al., 2000b; Primavera and de la Pena, 2000; Glaser, 2003
Wood fiber for rayon, paper	Christensen, 1982; FAO, 1985; Bhattacharyya, 1990; Ong, 1995; Bandaranayake, 1998; Ewel et al., 1998a
Buds and leaves for vegetables, alcohol, livestock fodder	Morton, 1965; Walsh, 1977; Christensen, 1983b; Semesi, 1998; Dahdouh-Guebas et al., 2006a; Jayatissa et al., 2006
Plant parts and extracts for medicines, pesticides	Sangdee, 1986; Chang and Peng, 1987; Bandaranayake, 1998, 2002; Sánchez et al., 2001; Primavera et al., 2004
Habitat for collecting honey, bees wax, and hunting wildlife	Hamilton and Snedaker, 1984; Untawale, 1987; Adegbehin, 1993; FAO, 1994; Basit, 1995; Sathirathai and Barbier, 2001; Nagelkerken et al., 2008

species for less favored ones – or even non-mangrove species – especially where the preferred wood has become less available or too costly to obtain (Walters, 2003). Harvest for fuelwood is often non-selective: some species are clearly better than others, especially for making charcoal, but evidence suggests people will harvest and burn as fuelwood almost any type of mangrove tree and are more likely to make decisions about which ones to harvest based on relative availability, rather than species preference (Walters, 2005a). In short, the material poverty of coastal communities and their widespread dependence on mangrove wood products to meet basic subsistence needs means users are often not in a good position to be selective and, instead, will harvest what is most readily available to them (Ewel et al., 1998a).

Patterns of harvest reflect the spatial distribution and relative accessibility of mangroves, which varies depending on local geomorphology and hydrology, socio-economic conditions, and past human disturbance (Ewel et al., 1998a; Hauff et al., 2006; Walters, 2003). Small-block clear-felling is applied, but to a limited extent and usually only in intensively managed forests (Hussain, 1995; Walters, 2004). Individual tree species vary dramatically in natural distribution within a mangrove and are often clumped in mono-specific stands. The dense above-ground root and branch growth of mangroves tends to make access to and clearing of forests difficult. These factors encourage the selective cutting of individual tree stems, branches and roots. To avoid such difficulties, pond construction in mangroves often starts with dike enclosures to retain water and kill the trees by flooding (for later clear-felling). It is also common for wood harvesting to concentrate on either the landward or seaward edges of a forest or along mangrove creeks, sites more readily accessible by foot during low tide or by boat during high tide (Walters, 2005a; Hauff et al., 2006; Lopez-Hoffman et al., 2006). Other things being equal, mangroves in proximity to human settlements are more likely to be heavily harvested. But whether and where mangroves are cut can also reflect the actions of government and coastal land owners who may restrict forest cutting. Yet, such restrictions may have limited effect on actual cutting practices given the practical difficulties of monitoring sites that are remote and simultaneously accessible by land and sea (Dahdouh-Guebas et al., 2000b, 2006a; Glaser, 2003; Walters, 2003, 2005a; Lopez-Hoffman et al., 2006).

Considerable research has been devoted to understanding the ecological effects of selection cutting and clear-felling as these treatments are applied in certain managed forests in Ecuador and South and Southeast Asia (Christensen, 1983a; FAO, 1985; Putz and Chan, 1986; Azariah et al., 1992; FAO, 1994; Nurkin, 1994; Blanchard and Prado, 1995; Hussain, 1995; Gong and Ong, 1995). But the relevance of this work is limited given that relatively little of the world's mangroves are subject to this kind of intensive forest management. In contrast, there has been remarkably little study of the ecological effects of informal, small-scale mangrove cutting by local coastal communities, a commonplace phenomenon that impacts mangroves in almost every region of the world.

Initial studies suggest that small-scale cutting typically involves the selective removal of one or few tree stems and/or branches at a time, causing localized structural disturbances that create relatively small gaps in the forest canopy (Smith and Berkes, 1993; Ewel et al., 1998b; Allen et al., 2001; Pinzon et al., 2003; Walters, 2005b). The creation of such gaps can alter micro-environmental conditions within the forest (Ewel et al., 1998b). Whereas clear-felling of mangroves tends to encourage regeneration of tree species that are better able to exploit large openings through seed dispersal and establishment, such as *Rhizophora* spp. and *Bruguiera* spp. (Putz and Chan, 1986; Blanchard and Prado, 1995; Hussain, 1995; Kairo et al., 2002; but see Azariah et al., 1992), the smaller openings created by selective cutting may better favor regeneration of species that successfully re-sprout/coppice from surviving stems, including *Sonneratia* spp., *Avicennia* spp., and *Laguncularia racemosa* (L.) Gaertn. f. (Smith and Berkes, 1993; Walters, 2005b; but see Pinzon et al., 2003). In contrast, the adult trees of *Rhizophora*, *Ceriops* and other genera of the Rhizophoraceae lack reserve meristems (Tomlinson, 1986), and therefore require replacement by new seedlings.

The cumulative effects of such selective cutting on a forest include reduced adult tree density, canopy height and canopy closure (Walters, 2005b; Hauff et al., 2006; Lopez-Hoffman et al., 2006). Heavily impacted stands are often characterized by few species of widely dispersed, dwarf-like trees manifesting a distinctly "bushy" appearance. Collateral damage from selective wood cutting may result in a net increase of dead wood in the forest (Allen et al., 2000). By contrast, local people in some settings intentionally forage for deadwood (for fuel) and thereby reduce levels of naturally-occurring deadwood (Walters, 2005a). These various changes in forest structure, composition and micro-climate can significantly alter the habitat conditions for establishment of seedlings (Bosire et al., 2003, 2006) and for resident marine and terrestrial animals (e.g., Barnes, 2001; Bosire et al., 2004, 2005a, b; Crona and Rönnbäck, 2005; Crona et al., 2006; Crona and Rönnbäck, 2007).

4. Mangrove-associated fisheries

4.1. Mangrove support functions to fisheries

Fishery species that use mangroves as habitat can be classified into permanent residents, spending their entire life cycle in mangrove systems, temporary long-term residents, associated with mangroves during at least one stage in their life cycle, and temporary short-term residents or sporadic users of the mangrove habitat (Robertson and Duke, 1990b). The critical early life stages, i.e. the larvae and juveniles, of many fish and shellfish species utilize mangroves as nursery grounds, whereafter they emigrate to other systems such as coral reefs as adults (Matthes and Kapetsky, 1988; Robertson and Duke, 1990a; Ogden, 1997; Barletta-Bergan et al., 2002a, b; Nagelkerken et al., 2002; Crona and Rönnbäck, 2007; Serafy and Araújo, 2007). Through the abundance of early life stages, mangroves also attract carnivorous fishes that conduct feeding migrations to mangrove areas.

The postlarvae of many commercial penaeid shrimps enter mangrove-dominated environments, where they develop into juveniles and subadults before migrating back to sea to complete their life cycle (e.g., Dall et al., 1990; Chong et al., 1990, 1996; Vance et al., 1996; Primavera, 1998b; Rönnbäck et al., 1999, 2002). Mangrove mud crabs, sergestid shrimps, and giant freshwater prawn are other crustaceans of commercial value that utilize mangroves as habitat during some life stage. Highly valued food and game fish that have a close association with mangroves include groupers, snappers, sea-perch, mullets, catfishes, milkfish, and tarpons. Mangroves also support many mollusk species that constitute an important in situ fishery. Edible species of oysters, mussels, cockles, and gastropods are collected extensively for local consumption, usually by the families of local fishermen, and/or market sale, e.g., the mangrove clam *Anodontia edentula* Linn. (Primavera et al., 2002). For more detailed information on fish and invertebrates associated with mangrove environments see Macintosh (1982), Rönnbäck (1999), and the biogeographic analysis by Matthes and Kapetsky (1988).

Mangroves also indirectly support fisheries where the harvested species never enter mangrove environments. Mangroves, seagrass beds, unvegetated shallows, and coral reefs can exist in isolation from each other, but commonly form integrated ecosystems of high productivity (Yanez-Arancibia et al., 1993; Ogden, 1997; Rönnbäck, 1999). For example, the ability of mangroves to control water quality (trapping and assimilating sediment and nutrients) is a prerequisite for coral reef functioning, including fisheries production (Kühlmann, 1988).

Another indirect support function to fisheries is the bioeconomics of shrimp trawling. Penaeid shrimps, which dominate global shrimp catches, are one of the most important fishery resources worldwide in terms of volume of catch and value per unit catch (Dall et al., 1990). Because penaeid shrimp sales generate most of the revenues from mechanized trawling in developing countries, shrimps (and indirectly their nursery habitat, i.e. mangroves) effectively subsidize commercial fish harvesting efforts by these vessels, including fish species not using mangroves as habitat (Turner, 1977; Bennett and Reynolds, 1993; Rönnbäck, 1999). Trawl catch ratio between marketed fish and penaeids in Indonesia was 667 kg of fish for every 100 kg of shrimps trawled (Turner, 1977).

Apart from fisheries aimed directly for human consumption, mangroves also support aquaculture operations by providing seed, broodstock and feed inputs (Rönnbäck, 1999; Naylor et al., 2000). Mangroves function as nursery grounds for the early life stages of aquaculture species like penaeid shrimps, mangrove mudcrabs, sea-perch, snapper, grouper, milkfish, etc. (Matthes and Kapetsky, 1988; Bagarinao, 1994; Primavera, 1998b; Walton et al., 2006a; Cannicci et al., 2008; Nagelkerken et al., 2008). The collection of wild seed, which supports major fishery operations in many countries, has however been criticized for bycatch problems. For example, the tiger prawn (*Penaeus monodon* Fabricius), which dominates shrimp aquaculture production, constitutes a very small proportion (down to 0.1%) of fish and invertebrate larvae in seed collector's catch (reviewed by Primavera, 1998a). This bycatch is usually sorted out on land and not returned to the sea, which could have significant negative impacts on biodiversity and capture fisheries production in the area. Some countries have developed hatcheries for seed production of cultured species. This may have reduced the dependence on mangroves to produce wild seed, but has increased demand for wild-caught broodstock instead. For instance, penaeid shrimp hatcheries often rely on the continuous input of mature females to sustain productivity as well as to avoid inbreeding problems. The mangroves in the Godavari delta, India, have been estimated to support an annual catch around 50,000 tiger prawn (*Penaeus monodon*) spawners, valued at US$ 6 million (Rönnbäck et al., 2003).

Mangroves and aquaculture are not necessarily incompatible. Already, the culture of seaweeds, mollusks and fish in cages in subtidal waterways is both compatible with mangroves and amenable to small-scale, family-level operations (Primavera, 1993, 1995). But there remains a need for mangrove-friendly aquaculture technology in the intertidal forest or swamp that does not require clearing of the trees. Development of such technology is on two levels: (a) silvofisheries or aquasilviculture where the low-density culture of crabs and fish is integrated with mangroves and (b) mangrove filters where adjacent mangrove stands are used to absorb effluents from high-density shrimp and fish culture ponds (Primavera, 2000b; Primavera et al., 2007). Present-day versions of integrated forestry–fisheries–aquaculture can be found in the traditional *gei wai* ponds in Hong Kong, mangrove–shrimp ponds in Vietnam, aquasilviculture in the Philippines, and silvofisheries in Indonesia (Primavera, 2000b). The Southeast Asian Fisheries Development Center Aquaculture Department has recently put out guidelines for sustainable aquaculture in mangrove ecosystems (Bagarinao and Primavera, 2005).

4.2. Economic importance of mangrove-associated fisheries

Fisheries production constitutes the major value of marketed natural resources from mangrove ecosystems. In terms of habitat use, the mangrove support to commercial, recreational and subsistence fisheries is well documented (see review in Rönnbäck, 1999). For instance, 80% of all marine species of commercial or recreational value in Florida, USA, have been estimated to depend upon mangrove estuarine areas for at least some stage in their life cycles (Hamilton and Snedaker, 1984). The relative contribution of mangrove-related species to total fisheries catch can also be significant, constituting 67% of the entire commercial catch in eastern Australia (Hamilton and Snedaker, 1984), 49% of the demersal fish resources in the southern Malacca Strait (Macintosh, 1982), 30% of the fish catch and almost 100% of shrimp catch in ASEAN countries (Singh et al., 1994).

Non-marketed catch is never included in fishery statistics, although coastal subsistence economies in many developing countries harvest substantial amounts of fish and shellfish from mangroves (Fig. 1F). The contribution of subsistence fisheries to total catch supported by mangroves was estimated at 10–20% in Sarawak (Bennett and Reynolds, 1993), 56% in Fiji (Lal, 1990), and 90% in Kosrae (Naylor and Drew, 1998). The annual subsistence harvest per household has been valued at US$610 in Fiji (Lal, 1990) and $900 in Irian Jaya, Indonesia (Ruitenbeek, 1994). For the poorest coastal families, mangrove fisheries clearly have an emergency food provision function and constitute the main source of protein in their diet (Magalhaes et al., 2007).

The most frequently used method to assess the mangrove support to commercial fisheries is the production function approach, where mangroves are put in as a determinant for fisheries catch (Barbier, 1994, 2003). Positive correlations between offshore yield of penaeid shrimps and amount of mangrove forest in the nursery area have been demonstrated throughout the tropics (e.g., Turner, 1977; Pauly and Ingles, 1986; Baran and Hambrey, 1998; Lee, 2004), whereas studies on other crustaceans, fish and molluscs are scarce (Rönnbäck, 1999). Correlations have been found between penaeid catches and latitude (inversely proportional) by Turner (1977) and Pauly and Ingles (1986), and with extent of intertidal areas and tidal amplitude (Lee, 2004). Furthermore, Pauly and Ingles (1986) found a non-linear logarithmic relationship between mangrove area and penaeid shrimp production, implying that the shrimp fisheries impact of reducing mangrove area becomes greater as the remaining area is reduced. Similarly, the length of mangrove-lined estuary or habitat edge where juvenile prawns have access to the mangrove is a more important indicator of shrimp densities than total area per se (Staples et al., 1985; Chong, 2007).

Quantitative estimates of fisheries production supported by mangroves have mainly focused on penaeid shrimps (e.g., Christensen, 1982; Lal, 1990; Ruitenbeek, 1994; Barbier and Strand, 1998), and there is a severe lack of productivity and monetary estimates for other fisheries (Nickerson, 1999; Rönnbäck, 1999). This may be related to the varying degree of mangrove importance as nurseries for fish, especially in the presence of alternative habitats like seagrass beds (Robertson and Duke, 1990a; Nagelkerken et al., 2000, 2002; Nagelkerken and van der Velde, 2004). To identify and value total commercial and subsistence fisheries catch supported by mangroves, economic analyses must take into account: (1) the large number of resident and transient species that utilize mangroves as habitat; (2) the biophysical interactions in the coastal seascape biome; (3) the direct and indirect subsidies of shrimp trawlers and mangroves, respectively, to total fisheries catch; and (4) the aquaculture industry's dependence on inputs like seed, broodstock and feed (Rönnbäck, 1999). By acknowledging these support functions, the potential life-support value of mangroves to fisheries is in the order of 1–10 tons of fish and shellfish per ha and year (first sale value ≈ 1000–10,000 US$ in developing countries) (Rönnbäck, 1999).

5. Mangrove ecosystem services

Mangroves support a wide variety of ecosystem services (e.g., Saenger et al., 1983; Ewel et al., 1998a; Moberg and Rönnbäck, 2003; Barbier, 2007; Rönnbäck et al., 2007a), which can be classified into supporting, provisioning, regulating and cultural services (Millennium Ecosystem Assessment, 2005). Supporting services are those that are necessary for all other ecosystem services, and include soil formation, photosynthesis, primary production, nutrient cycling and water cycling. Provisioning services are the natural products generated by mangroves (see previous sections).

Regulating ecosystem services are the benefits obtained from the regulation of ecosystem processes such as resilience, pollination, biological control, nutrient cycling, air quality regulation, and maintenance of biodiversity for ecosystem function and resilience, etc. (Millennium Ecosystem Assessment, 2005; Rönnbäck et al., 2007b; Bosire et al., 2008; Cannicci et al., 2008; Gilman et al., 2008; Kristensen et al., 2008; Nagelkerken et al., 2008). Regulating services analyzed in detail below include water quality maintenance, environmental disturbance prevention (storm, flood and erosion control) and climate regulation. One critical function supporting all these services is that mangroves effectively retard water flow, mainly as a function of the trees' three-dimensional structural complexity and the complex topographical features of channels, creeks, etc. This enables efficient trapping of suspended and particulate matter, which can lead to land accretion buffering against potential sea level rise in the future.

Favorable sediment characteristics and high photosynthetic rates of many mangrove systems provide the basis for the biofilter function with high nutrient uptake levels (Rivera-Monroy et al., 1995; Robertson and Phillips, 1995; Alongi et al., 2000). Peri-urban coastal areas of the developing world receive extensive amounts of untreated sewage, and mangroves certainly filter this discharged wastewater, thereby limiting coastal sewage pollution. Based on the cost of constructing a sewage treatment plant, the value of biofilter functions of mangroves has been estimated at US$ 1193 ha^{-1} $year^{-1}$ to US$ 5820 ha^{-1} $year^{-1}$ depending on types and extent of mangroves (Table 2). The wide-scale conversion of mangroves to accommodate shrimp farms removes the natural biofilter function of surrounding mangroves. Consequently, waste laden pond effluent water is reused causing self-pollution (Rönnbäck, 1999; Kautsky et al., 2000) in the farm system itself, but also affecting remaining mangroves and littoral habitats, often of primary importance for collection of marine products by local communities. Robertson and Phillips (1995) estimated that up to 22 ha of mangrove forest would be required to filter the nutrient load per hectare of intensive shrimp pond. More recently, Primavera et al. (2007) showed that 1.8–5.4 ha of mangroves are required to remove nitrates in effluents from 1 ha of shrimp pond.

Mangroves are considered as a natural barrier protecting the lives and property of coastal communities from storms and cyclones, flooding, and coastal soil erosion (Farber, 1987; Othman, 1994; Sathirathai and Barbier, 2001; Lal, 2002; Walters, 2003, 2004; Badola and Hussain, 2005; Hong, 2006; Barbier, 2007). Values ascribed to this service include, for example, US$ 120 per household (Badola and Hussain, 2005), and US$ 3700 ha^{-1} (Sathirathai and Barbier, 2001) and US$ 4700 ha^{-1} (Costanza et al., 1989) of mangrove (Table 2). These are major indirect benefits and a principal reason for planting mangroves along many low-lying coasts. Artificial structures to replace the coastal protection services provided by mangroves can be expensive (Moberg and Rönnbäck, 2003; Walters, 2003) and may not be as effective (Badola and Hussain, 2005; Barbier, 2006).

In particular, the Indian Ocean Tsunami disaster of December 26, 2004, which killed over 200,000 people and damaged livelihoods and coastal resources in 14 Asian and African countries, highlighted the role of protection and sound management of the coastal environment and provided a stark reminder that environmental sustainability and human security are inseparable (Walters, 2006).

The tsunami disaster has received scientific and media attention worldwide, and the protective function of mangroves for landward human settlements has been often highlighted. Yet, most reports with respect to protection by mangrove forests were either very localized and/or anecdotal in nature (Danielsen et al., 2005; Harakunarak and Aksornkoae, 2005; IUCN, 2005; Liu et al., 2005; Roy and Krishnan, 2005; Williams, 2005; Dahdouh-Guebas, 2006; Stone, 2006; Wells and Kapos, 2006). This has prompted two, contradicting 'narratives' among authors and policy-makers regarding the protective role of mangroves. On one hand, some have generalised the protective function of mangroves as documented from some areas to entire coastlines and countries and therefore over-interpreted the role of mangroves. On the other hand, others have generalised the apocalyptical nature of a tsunami based on the Banda Aceh experience and minimalised the role of mangroves to the extent of suggesting that they are ineffective and that more effort should be focused on tsunami alert systems (Overdorf and Unmacht, 2005; Baird, 2006). Both views have been criticized because of insufficient examination of results or assumptions supporting this function (Dahdouh-Guebas et al., 2005c; Kathiresan and Rajendran, 2005; Dahdouh-Guebas and Koedam, 2006).

The role of mangroves in wave attenuation has long been scientifically proven (Furukawa et al., 1997; Wolanski, 1995; Mazda et al., 1997; Massel et al., 1999). Reduction of waves depends on water depth, wave period and height, quality of the mangrove forest, and type of aerial root systems (Mazda et al., 1997; Kathiresan, 2003; Dahdouh-Guebas et al., 2005c). The post tsunami studies have found that human deaths and loss of property was a function of type and area of the coastal vegetation shielding the villages (Dahdouh-Guebas et al., 2005c; Kathiresan and Rajendran, 2005; but see Kerr and Baird, 2007). Further evidence of the storm protective value of mangroves can be found in studies of local peoples' knowledge and practices. Among some coastal communities in the Philippines and India there is a widely-held appreciation for the storm protective function of mangroves, and many people plant and protect mangrove trees explicitly for this purpose (Fig. 1A; Walters, 2003, 2004; Badola and Hussain, 2005; Walton et al., 2006b). It is common practice for small-boat fishers in these countries to seek the shelter of mangroves during storms, but sheltering in deep mangrove creeks also provided protection to commercial, recreational and naval vessels in the port of Cairns, Australia when tropical cyclone Larry crossed the Queensland coast on 20 March 2006 (Williams et al., 2007). Some earlier studies have also suggested that the loss of lives due to hurricanes, tidal waves, typhoons, etc. could have been reduced by the presence of a mangrove protective belt (Fosberg, 1971; Primavera, 1995; Mazda et al., 1997; Massel et al., 1999).

Mangrove ecosystems are among the most productive and biogeochemically active ecosystems and represent potentially important sinks of carbon in the biosphere (Twilley et al., 1992; Ong, 1993; Gattuso et al., 1998). Clough et al. (1997) calculated net photosynthetic rates of 155 kg C ha^{-1} per day in a 22-year old *Rhizophora apiculata* Bl. forest in Malaysia (Table 2). The carbon stock per unit area can also be enormous as the top layers of mangrove sediments store large amounts of organic carbon, typically an order of magnitude higher than those of other tropical forests. Successful management of mangrove ecosystems thus has the potential to produce a 'measurable' gain in CO_2 sequestration (Ayukai, 1998), a characteristic likely to acquire greater attention with the forecasted global warming this century.

Cultural services stem from dynamic and complex social attributes. The variety within coastal ecosystems provides humans with almost unlimited opportunities for aesthetic and recreational experiences, cultural and artistic inspiration, as well as spiritual and religious enrichment (Fig. 1B; Mastaller, 1997; Kaplowitz, 2001; Rist and Dahdouh-Guebas, 2006; Rönnbäck et al., 2007b). An intriguing illustration comes from the *Asmat* from Irian Jaya, Indonesia, who have largely preserved their traditions and beliefs (Mastaller, 1997). According to their legends, their creator carved human-like figurines out of a mangrove root which came to life when he played a self-made drum out of a mangrove tree (loc. cit.). Today, *Rhizophora* roots are still used to carve mystic totem poles (loc. cit.).

The location of mangroves along the coastline, often proximate to populated areas, combined with their unique ecological and aesthetic character, affords opportunities for development of eco-tourism and environmental education. Many coastal communities have co-evolved with their local mangrove ecosystems. Their traditional use of mangrove resources is often intimately connected with the health and functioning of the system. These uses are often governed by customary rights, traditions and heritage, and they are often closely tied to the culture of the local communities. The failure to recognize these customary use rights has often resulted in the alienation of local communities in managing local mangrove ecosystems, and in participating in the replanting and rehabilitation of mangroves (Walters, 2004; Barbier, 2006), subsequently undermining incentives for, and use of, LEK which could be valuable for management purposes.

6. Mangrove management, planning and policy

6.1. Property rights, resource access and conflict

Mangroves are unusual environments in that they are located between dry land and shallow marine and brackish water. This characteristic introduces complexities to planning and management because of competing and overlapping interests in mangrove lands and their resources. In short, mangroves are valuable coastal lands to various forest users and land developers, each one having incentive to claim and control access through degrees of privatization. But this tenure dynamic changes because marine and estuarine waters in mangroves as elsewhere are typically viewed as open access transportation corridors for fishing boats, and the diverse fish and crustaceans within these waters are usually treated as a common property resource available for harvest by local fishermen.

These complexities are often mirrored in government policy. Until recently, most governments considered mangroves to be relatively worthless swamplands, so rational policy guiding their management has in most cases been late in coming. Being part land and part sea, jurisdictional ambiguities are often present. For example, regulation of mangrove forest lands in the Philippines has historically fallen under the legal jurisdiction of both the Department of Environment and Natural Resources (formerly the Ministry of Forests), whose mandate was to protect and sustainably manage these as forests, and the Department of Agriculture, whose mandate was to promote brackish water aquaculture development in these same areas (Primavera, 2000a, 2005; Walters, 2003). Thus, government decisions concerning mangroves were often made with "...the right hand not knowing what the left hand was doing" (Primavera, 1993, p. 168). Similar problems of jurisdictional ambiguity over mangroves have been documented in Ecuador (Meltzoff and LiPuma, 1986), India (Bhatta and Bhat, 1998; Dahdouh-Guebas et al., 2006a), Thailand (Vandergeest et al., 1999), Sri Lanka (Dahdouh-Guebas et al., 2000a, b), Indonesia (Armitage, 2002) and Brazil (Glaser and Oliveira, 2004).

But such ambiguities go beyond government policy and affect informal understandings and customary rules concerning access and use of mangroves by different users. Customary use of mangroves is typically characterized by common access rights, with different uses overlapping but to a large degree accommodating one another (Fig. 1E; Bhatta and Bhat, 1998; Walters, 2004). Conflict in such situations can arise, for example, where customary boat access or seine fishing rights become impaired by the construction of a dyke or the planting of mangrove trees (Walters, 2004), or where resident mangrove fishers and wood users are forced to compete with outsiders for the same resources (Glaser and Oliveira, 2004). The potential for such conflict is exacerbated where large tracts of mangrove are leased to private interests who displace common access users (Bailey, 1988; Dewalt et al., 1996; Stonich and Bailey, 2000; Walters, 2003, 2004; Hoq, 2007). The issue of shrimp farming is particularly problematic because the large profit potential of these operations creates incentive for corruption of legal mechanisms that might otherwise protect the forests and/or interests of local users (Meltzoff and LiPuma, 1986; Bhatta and Bhat, 1998; Stonich and Vandergeest, 2001; Armitage, 2002; Dahdouh-Guebas et al., 2002). In short, conflict is more likely to emerge in the absence of shared understandings about rules of access, clear government regulations, and effective means of enforcement and dispute resolution.

6.2. Deforestation and competing land uses

Mangrove forests are among the most threatened global ecosystems, especially in Asia, and current mangrove area has fallen below 15 million hectares, down from 19.8 million ha in 1980 (Wilkie and Fortuna, 2003). Global rates of loss in the past two decades vary from 20% (Wilkie and Fortuna, 2003) to 35% (Valiela et al., 2001). The average rate of 1.52% mangroves lost per year (Valiela et al., 2001; Alongi, 2002) shows an improvement from 1.9% in the 1980s to 1.1% in the 1990s (Wilkie and Fortuna,

Table 2
Examples of economic assessments of some regulating ecosystem services supported by mangroves

Regulating service	Values and benefits	Reference
Water quality maintenance (biofilter function)	US$ 5820 ha^{-1} $year^{-1}$	Lal, 1990
	US$ 1193 ha^{-1} $year^{-1}$	Cabrera et al., 1998
	7.4 and 21.6 ha of mangroves needed to remove nitrate and phosphorous, respectively, in effluents per ha of intensive shrimp pond	Robertson and Phillips, 1995
	1.8–5.4 ha of mangroves needed to remove nitrate in effluents per ha of shrimp pond	Primavera et al., 2007
Environmental disturbance prevention (storm, flood and erosion control)	US$ 4700 ha^{-1}	Costanza et al., 1989
	US$ 3679 ha^{-1}	Sathirathai and Barbier, 2001
	US$ 120 per household	Badola and Hussain, 2005
Carbon sink	155 kg C ha^{-1} day^{-1}	Clough et al., 1997
	1500 kg C ha^{-1}	Ong, 1993

2003). Nevertheless, the prospect of a world without mangroves appears to be real (Duke et al., 2007). Although many factors are behind global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of brackish water fish and shrimp farms (Primavera, 1995; Barbier and Cox, 2003). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38% of mangrove deforestation; in Asia, aquaculture contributes 58% to mangrove loss with shrimp farming accounting for 41% of total deforestation (see Table 3 in Valiela et al., 2001). Other factors in mangrove decline are forest use, mainly for industrial lumber and woodchip operations (26%), freshwater diversion (11%), and reclamation of land for other uses (5%). The remaining causes of mangrove deforestation are herbicide impacts, agriculture, salt ponds and other coastal developments. A global survey of 38 coastal, island and estuarine mangrove stands confirmed that clear cutting and reclamation for agriculture and aquaculture, urban expansion and resort development threatened the majority (55%) of all sites visited (Farnsworth and Ellison, 1997).

The conversion of mangroves to aquaculture ponds has been fuelled by governmental support, private sector investment and external assistance from multilateral development agencies such as the World Bank and Asian Development Bank (Siddall et al., 1985; Verheugt et al., 1991). To quote a report of the 1978 Aquaculture Project in Thailand "The subproject will involve the large-scale development of mangrove swamps into small shrimp/fish pond holdings . . ." (ADB, 1978 in Primavera, 1998a). From US $368 million (representing only 14.1% of total fisheries assistance) in 1978–1984, international aid to aquaculture increased to $910 million (33.7% of total fisheries assistance) in 1988–1993 (Primavera, 1998a). The Asian Development Bank alone provided total aid to fisheries and aquaculture of $1085 million in the 1969–1996 period, including US $21.8 million in aquaculture loans for shrimp and milkfish ponds and hatcheries in the Philippines (Primavera, 1998a, 2000b). But the much earlier fishpond boom of the 1950s was fuelled by a loan of US$ 23.6 million for fishpond construction and operations from the International Bank for Reconstruction and Development intended "to accelerate . . . the conversion of vast areas of marshy lands [mangroves] . . . into productive fishponds" (Villaluz, 1953, in Primavera, 2000a).

The effects of this decline in mangrove area are exacerbated by the widespread degradation of remaining forests, the result of over-cutting of wood and over-harvesting of mangrove aquatic resources. The extent of such degradation is not well documented, but case studies reveal dramatic changes to the structure and composition of harvested forests and associated declines in resource availability to local communities (Kairo et al., 2002; Walters, 2005b). Infrastructure developments and upland land use can cause sedimentation and changes to hydrology that impact mangroves at some distance, causing the gradual die-back of particular species or entire stands (Dahdouh-Guebas et al., 2005b). Ironically, such ecological degradation can be masked by the expansion of less typical, less functional and less vulnerable species and thus take the form of 'cryptic ecological degradation' (sensu Dahdouh-Guebas et al., 2005b).

Problems of deforestation and degradation are compounded by growing human populations in many coastal areas (Primavera, 2000a). The Philippines offers a case in point: mangroves once abundant around Manila Bay at the turn of the last century have since been entirely cleared, the combined result of fish pond development, urban infrastructure expansion and residential spread (Brown and Fischer, 1918; Cabahug et al., 1986). Similarly, in a more rural region of the country, Bais Bay, mangroves have declined in area over the past 50 years by 75% at the same time that coastal populations have increased 10-fold (Walters, 2003). Population growth coinciding with declining mangrove area has likewise been documented along the coastlines of Honduras (Dewalt et al., 1996), Vietnam (de Graaf and Xuan, 1998) and Bangladesh (Bashirullah et al., 1989).

6.3. *Mangrove silviculture*

Mangrove silviculture has been practiced in some Asian countries since the 19th century (Brown and Fischer, 1918; Watson, 1928; Curtis, 1933; Hussain and Ahmed, 1994; Kaly and Jones, 1998; Vannucci, 2002). Mangroves are planted for various purposes, including (i) wood production to support commercial or small-scale forestry; (ii) shoreline protection, channel stabilization and storm protection for coastal human settlements from cyclones and other extreme natural events, and for protection against seawater intrusion; (iii) fisheries, aquaculture and wildlife enhancement; (iv) legislative compliance with protective measures and compensatory requirements; (v) social enrichment (e.g., aesthetics, income generation through eco-tourism); and (vi) ecological restoration (Field, 1996; Bhatta and Bhat, 1998; Kairo et al., 2001; Walters, 2004; Walters et al., 2005). Nursery and planting techniques vary considerably among mangrove species, and the silvicultural methods chosen will depend on which of the above objectives are desired (Field, 1998; Saenger, 2002).

Traditionally, both clear-felling and selection systems have been used, and in some areas a mixed system has been employed (FAO, 1994). Clear-felling systems applied to mangrove forests are the most cost-effective, although erosion and site deterioration risks as well as the loss of ecosystem services are higher. Clear-felling has been found suitable for some economically valuable species, such as *Rhizophora apiculata*, *R. mucronata* Lamk. and

R. stylosa Griff., which are strong and light-demanding and so can withstand competition in open areas. In selection systems, the stands are uneven-aged and the forest cover is never completely removed. They are more environment-friendly since marketable trees are harvested periodically and over all parts of the forests, providing better soil protection and biodiversity, reducing risks of insect damage and invasions, and offering improved wind buffering. However, selection systems are less cost-effective due to their complexity and greater labor requirements.

Mangrove silvicultural practices have produced mixed results depending on the practices. For example, the success of mangrove management since the beginning of the 20th century in Matang, Malaysia is mainly due to intensive reforestation efforts (Ong, 1995; Chan, 1996), although decline in yields has been reported since the late 1960s (Gong et al., 1980; Gong and Ong, 1995). Likewise, multi-use managed forests in the Sunderbans have maintained long-term productivity through the application of scientific silvicultural practices with traditional knowledge (Vannucci, 2002). In Venezuela, however, the Guarapiche Forest Reserve, San Juan River is yet to recover fully despite well-planned silvicultural practices (Lacerda et al., 2002). Although restored mangrove forests may resemble forest plantations rather than natural forests, such plantations can be a first step toward mangrove rehabilitation (Ellison, 2000; Bosire et al., 2003; Bosire et al., 2008; but see Walters, 2000). To improve the success in rehabilitation, other silvicultural methods have been employed including natural regeneration, assisted regeneration and macro-propagation.

Reforestation of mangrove forests through natural regeneration is relatively inexpensive and maintenance is less labor-intensive. Natural regeneration leads to better early root development and causes less soil disturbance. However, the success of natural regeneration will depend on the state of degradation of the original mangrove. Although assisted regeneration is more expensive, its costs will vary depending on labor costs, site characteristics, proximity to propagule sources, and whether propagules, seedlings or transplants are used (Saenger, 1996). Assisted regeneration may be required at sites with insufficient natural regeneration. Approaches for macro-propagation of mangroves include direct planting of propagules collected from the wild, out-planting of up to 1-year-old nursery-raised propagules, direct transplanting of seedlings and shrubs, out-planting after nursery-raising small seedlings collected from the wild, raising of air-layered material, and use of stem cuttings (Carlton and Moffler, 1978; Hamilton and Snedaker, 1984; Field, 1996).

6.4. Ecological restoration

Ecosystem restoration to the original pristine state, or rehabilitation to recover some ecosystem functions, may be appropriate when a mangrove ecosystem has been altered so that normal processes of secondary succession or natural recovery from damage are inhibited in some way. Mangrove restoration is increasingly practiced in many parts of the world (Ellison, 2000; Kairo et al., 2001; Vannucci, 2002). Mangrove forests have been rehabilitated to achieve a variety of goals, e.g., for commercial purposes (Watson, 1928), restoring fisheries and wildlife habitat (Lewis, 1992; Stevenson et al., 1999), multiple community use purposes, or shoreline protection purposes (Thorhaug, 1990; Saenger and Siddiqi, 1993; Bhatta and Bhat, 1998; Field, 1998; Walters, 2004; Barbier, 2006; Walton et al., 2006b).

There is already a great deal of knowledge and experience in rehabilitating mangroves by artificial means around the world (Field, 1996, 1998). However, many of these efforts are carried out without considering the experience and lessons learned from similar projects, resulting in duplication of efforts and waste of resources (Elster, 2000; Kairo et al., 2001). Recently, interest has focused on indigenous or folk technologies for mangrove restoration. For example, local fisherfolk have been planting mangroves in some areas of Southeast Asia for decades, well before governments and non-government organizations began to promote the activity as a conservation tool (Fig. 1E; Fong, 1992; Weinstock, 1994; Walters, 2000, 2004). These local management systems are relatively small-scale and utilize simple technologies, but they can be rich in knowledge and practical experience that is usually overlooked by "experts" who promote mangrove reforestation (Vayda et al., 2004; Walters, 1997; Walters et al., 2005).

Failure to better understand the local environmental and socio-economic contexts of mangrove restoration dooms many such efforts. Mangrove restoration projects often have moved immediately into planting of mangroves without determining the cause of previous degradation or why natural recovery has failed (Lewis, 2000, 2005). Even where environmental conditions permit natural or assisted restoration of a site, ongoing or future disturbance of the area by local people may prevent it (Walters, 1997). Ideally, mangrove restoration success should be measured as the degree to which the functional replacement of natural ecosystem has been achieved. However, long-term success in mangrove replanting will be determined by the level of support and involvement of local communities and local governments (Primavera and Agbayani, 1997; Walters, 1997, 2004; Lewis, 2000; Barbier, 2006). Mangrove rehabilitation programs that only utilize coastal communities as sources of replanting labor and do not involve them in the long-run management of the various uses of the restored ecosystem are less likely to be successful (Rönnbäck et al., 2007a).

A review of mangrove (re)planting in the Philippines over the past century shows a change from community-led efforts to projects externally driven by international development grants and loans. This change in drivers is paralleled by an increase in planting costs from <$100 ha^{-1} to over $500 ha^{-1}, yet long-term survival rates generally remain low. Poor survival can be traced to inappropriate species (*Rhizophora* is favored over the natural colonizers *Avicennia* and *Sonneratia* because it is easier to plant), and unsuitable sites in open access but suboptimal lower intertidal to subtidal zones, rather than the ideal but contentious middle to upper intertidal areas which have long been converted to aquaculture ponds. For mangrove rehabilitation efforts to succeed, funding appears to be of secondary importance relative to suitable sites and species, community involvement and commitment, and grant of tenure.

6.5. Geo-spatial monitoring and analysis

In order to develop and implement effective policy regarding the socio-economic use of mangrove forests, it is essential that stakeholders have access to accurate and cost-effective techniques for mapping and monitoring these coastal wetlands. Given that many of these forests are quite large, are located in remote areas and have been experiencing rapid changes, it is not surprising that various remote sensing techniques have been employed to determine their spatial distribution and health. Traditional aerial photography is still being employed (e.g., Krause et al., 2004; Dahdouh-Guebas et al., 2006b) to map these forests, but given their repetitive coverage with constant image quality and immediate ease of operation, the use of satellite imagery, both optical and radar, now govern this endeavor. Satellite imagery enables resource managers to quickly map and continuously monitor their mangroves without the constant need for exhaustive field surveys. Using very high resolution imagery, the development of single species or even trees can be monitored, which may be necessary in light of selective cutting and ecological degradation (Dahdouh-Guebas et al., 2005a). Moreover, these digital data are easily transferable into Geographic Information Systems for spatial analyses studies at a broader coastal management level.

There are two types of space-borne data available for mangrove forest mapping, optical and radar. Optical sensors rely on reflected sunlight, primarily in the visible and infra-red regions of the electromagnetic spectrum. With regards to mangroves, the signals received can provide information regarding the photosynthetic activity of the trees which can then be used to distinguish them from other non-mangrove land covers or even between mangrove species or mangrove conditions (e.g., unhealthy stands). Conversely, Synthetic Aperture Radar (SAR) satellites actively emit microwave energy to their targets. The returning radar signals from the surface (i.e. backscatter) are very sensitive to dielectric and geometric properties of mangrove canopies and can thus also be used as an alternative or supplement to optical mapping procedures.

To date the vast majority of investigations using space-borne platforms to map and monitor mangroves have focused on optical sensors, primarily from the traditional/conventional SPOT and Landsat satellite series. These satellites have been used to map mangroves in a myriad of countries including, for example, Australia (Long and Skewes, 1996), Brazil (Brondizio et al., 1996), New Zealand (Gao, 1998), Thailand (Webb et al., 2000), the Turks and Caicos Islands (Green et al., 1998), the United Arab Emirates (Saito et al., 2003) and Vietnam (Tong et al., 2004). In comparison to the recent launch of very high resolution optical satellites (e.g., IKONOS in 1999), these traditional sensors are limited in spatial resolution (e.g., $\sim$1 m versus $\sim$25 m pixel size). However, these satellite data are cheaper, provide a larger coverage per acquisition, are easier to process and have extensive records (e.g., Landsat data extending back to 1972).

Consequently, they continue to play a very crucial role in assessing historical changes in mangrove forests. For example, multi-temporal SPOT and multi-temporal Landsat images have been used to determine the rates of mangrove forest degradation occurring in Madagascar (Rasolofoharinoro et al., 1998) and Mexico (Kovacs et al., 2001a), respectively, both resulting from hydrologic modification incurred from channel projects. Rates of mangrove gradation and degradation resulting from natural cycles of coastal accretion and erosion have also been determined for the coast of French Guiana using multi-date SPOT satellite data (Fromard et al., 2004) and for the Para coastline (North Brazil) using multi-date Landsat data (Cohen and Lara, 2003). Multi-temporal satellite data have even been used to quantify the success of mangrove forest recovery resulting from the implementation government regulations on mangrove protection in Thailand (Muttitanon and Tripathi, 2005) and from very recent mangrove reforestation projects initiated by the Red Cross in Vietnam (Beland et al., 2006).

One major limitation to the use of the conventional sensors has been the inability to distinguish mangroves at the species level. In the aforementioned studies, mangroves are either simply separated from non-mangrove land cover/land use areas or they are further subdivided into 2–7 broad qualitative mangrove classes such as dense/tall or short/sparse mangroves. In a few circumstances, tall dense Rhizophora species have been mapped using Landsat data. Such mapping scales may suffice for many mangrove policy and management programs, especially in countries where only one species exists (e.g. New Zealand), but they could seriously hinder efforts where socio-economic policies on mangroves are based at the species level. Fortunately, studies in Panama (Wang et al., 2004a, b), Mexico (Kovacs et al., 2005) and Sri Lanka (Dahdouh-Guebas et al., 2005a) have shown that with the very high resolution optical satellites (IKONOS and Quickbird) mangroves can be accurately mapped at the species level from space.

Whilst the number of studies is extremely limited, researchers have shown that space-borne SAR can be used in conjunction with optical data or as an alternative in the mapping of mangroves (Aschbacher et al., 1995; Dwivedi et al., 1999; Kushwaha et al., 2000; Simard et al., 2002). The main advantages of SAR are that it is not limited to daylight and, most importantly, it can penetrate cloud cover. Consequently, in cloud persistent areas of the tropics, it may be the only viable method for mangrove monitoring. Moreover, depending on the polarization, incidence angle and wavelength, SAR can penetrate forest canopies providing additional information that is not possible from optical sensors. The studies of space-borne SAR have, to date, been limited to older SAR satellites which are limited not only in spatial resolution but in flexibility of incidence angle and polarization mode acquisition options. With the recent launch of a new generation of SAR satellites (e.g., C-band Radarsat-2, L-band ALOS Palsar), it is anticipated that, with their technological advancements (e.g., fully polarimetric capabilities), SAR mangrove mapping accuracies will dramatically improve.

Thus far, all of the studies cited have indicated that mangrove aerial extent can be mapped accurately from space and that these sensors can provide an effective method for long-term mangrove monitoring. However, in some circumstances, resource managers and policy-makers may require quantitative data (i.e., biophysical parameters) of their mangrove forests including measures of tree height, basal area, stem density and even biomass indicators such as Leaf Area Index (LAI) and allometric equations (cf. Komiyama et al., 2008). For example, they may wish to model the ecological response of a mangrove forest to hurricanes (Kovacs et al., 2001b) or determine how the biophysical parameters of their mangrove are modified by local cuttings (Walters, 2005b). Quantitative studies using remote sensing techniques require, initially, a significant amount of field data collection and are thus labor-intensive and expensive to conduct and possibly why so few of these studies are available.

With regards to conventional optical satellite data, significant relationships have been found between SPOT vegetation indices and both mangrove percent canopy closure (Jensen et al., 1991) and mangrove LAI (Green et al., 1997). Using simulated data, results from one study (Ramsey and Jensen, 1996) have also indicated that vegetation indices derived from Landsat and AVHRR data can also be correlated with mangrove LAI. More recently, significant relationships between mangrove LAI and IKONOS data have also been established (Kovacs et al., 2004a, b). Consequently, this parameter can now be estimated from optical satellite data at even the species level (Kovacs et al., 2005). As previously indicated, SAR can not only provide information on the geometry and water content of forest canopies but, in some circumstances, even collect data from below the canopy layer. For example, although using air-borne and not space-borne SAR, researchers (Mougin et al., 1999) in French Guiana have found not only significant relationships with radar backscatter and both mangrove height and biomass but also with mangrove stem density and basal area. With regards to space-borne SAR platforms, significant relationships have also been found between radar backscatter and mangrove LAI using both Radarsat-1 (Kovacs et al., 2006) and ENVISAT ASAR (Kovacs et al., 2008) satellite data. It is again anticipated that with the new generation of SAR satellites other mangrove forest biophysical parameter data could be extracted using radar backscatter signals.

Given the aforementioned advances in Earth observational imaging, it is no surprise that the availability of these data have significantly improved the ability of policy-makers and resource managers to monitor socio-economic impacts on their mangrove forests. Moreover, and possibly just as important, is the availability of these data to the general public. Specifically, satellite imagery, although in a limited format (e.g., limited spectral resolution), are now available on internet free access virtual globe programs such as Google Earth. In the hands of the public, these new tools could significantly alter the socio-economic dynamics associated with these forests at even the most local of scales.

7. Conclusions and future directions

Research on the human dimensions of mangrove forests remains a relatively new frontier. While not intended to provide a comprehensive list of possible research topics, these concluding comments suggest several key priorities.

There are a growing number of studies which examine local resource utilization and valuation of mangroves, yet coverage is patchy: limited to a relatively small number of sites, concentrated within a few biogeographic regions (esp. East Africa, Southeast Asia and the Indian subcontinent), and typically conducted over short time frames. Significant mangrove regions remain understudied (e.g., West Africa, South America, Indonesia). Furthermore, most of these studies exist in relative isolation from one another, yet opportunities to extract regional and global patterns are now warranted. Research that incorporates multi-year time frames and historical perspectives are particularly relevant given the rapid socio-economic and environmental changes unfolding along most tropical coastlines today. Likewise, there is need for economic valuation studies that explicitly focus on mangrove resources that are not marketed, but rather harvested and consumed directly by coastal households.

Studies that pay careful attention to the actual *ecology* of resource use are especially critical in light of the widespread influence of people on mangrove ecosystems (Walters, 2005b). Understanding how and why people actually harvest forest and aquatic resources in space and over time within a mangrove, and how these patterns of use impact the condition of the forest, is also vital for effective management, yet such information is almost always absent in planning and policy discussions. Standardised collection of this type of information from the local inhabitants is a first step in assuring that policy and law are anchored in local environmental and socio-economic reality (e.g., Kaplowitz, 2001; Omodei-Zorini et al., 2004; Walters, 2004; Dahdouh-Guebas et al., 2006a; Rist and Dahdouh-Guebas, 2006).

Location-specific studies should likewise be integrated with research that explicitly seeks to understand the range of human forces that impact mangroves less directly, but often more widely. Among these influences include (i) hydrological diversions caused by infrastructure developments along the coast or upstream of deltaic mangroves (e.g., dams); (ii) public policies with bearing on coastal natural resources, land use and development; (iii) markets for trade in mangrove products and products cultivated on former mangrove lands; and (iv) changes in sea level, rainfall and storm events associated with climate change.

The problems facing mangroves are dual: growing coastal populations put greater pressure on the ecosystem from the landward side, while global climate change, particularly sea-level rise, will increasingly put pressure on the mangrove from the seaward side. While the forest is squeezed as an ecosystem between these pressures, coastal subsistence users will be increasingly squeezed by economic pressures and public policies that respond to the same issues of overpopulation and global change. If resource management and land-use planning options to cope with these likely conditions are not effectively anticipated, both mangroves and the people who depend on them stand to lose.

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