

## Review

## Threats to sandy beach ecosystems: A review

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## ABSTRACT

We provide a brief synopsis of the unique physical and ecological attributes of sandy beach ecosystems and review the main anthropogenic pressures acting on the world's single largest type of open shoreline. Threats to beaches arise from a range of stressors which span a spectrum of impact scales from localised effects (e.g. trampling) to a truly global reach (e.g. sea-level rise). These pressures act at multiple temporal and spatial scales, translating into ecological impacts that are manifested across several dimensions in time and space so that today almost every beach on every coastline is threatened by human activities. Press disturbances (whatever the impact source involved) are becoming increasingly common, operating on time scales of years to decades. However, long-term data sets that describe either the natural dynamics of beach systems or the human impacts on beaches are scarce and fragmentary. A top priority is to implement long-term field experiments and monitoring programmes that quantify the dynamics of key ecological attributes on sandy beaches. Because of the inertia associated with global climate change and human population growth, no realistic management scenario will alleviate these threats in the short term. The immediate priority is to avoid further development of coastal areas likely to be directly impacted by retreating shorelines. There is also scope for improvement in experimental design to better distinguish natural variability from anthropogenic impacts. Sea-level rise and other effects of global warming are expected to intensify other anthropogenic pressures, and could cause unprecedented ecological impacts. The definition of the relevant scales of analysis, which will vary according to the magnitude of the impact and the organisational level under analysis, and the recognition of a physical–biological coupling at different scales, should be included in approaches to quantify impacts. Zoning strategies and marine reserves, which have not been widely implemented in sandy beaches, could be a key tool for biodiversity conservation and should also facilitate spillover effects into adjacent beach habitats. Setback and zoning strategies need to be enforced through legislation, and all relevant stakeholders should be included in the design, implementation and institutionalisation of these initiatives. New perspectives for rational management of sandy beaches require paradigm shifts, by including not only basic ecosystem principles, but also incentives for effective governance and sharing of management roles between government and local stakeholders.

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## 1. Introduction

The accelerating destruction of natural habitats and consumption of natural resources by rapidly expanding human populations has caused huge impacts to ecosystems across the globe. Amplified

by human population shifts towards the coast (Roberts and Hawkins, 1999), many of these impacts are focussed at the world's coastlines, which are dominated by sandy shores (McLachlan and Brown, 2006). Intense coastal development, the inevitable consequence of economic progress, has resulted in widespread modification of sandy beach ecosystems. Human changes to sandy shores began at least two centuries ago (Nordstrom, 2000), and are predicted to intensify over the next few decades (Brown et al., 2008). Global climate change, particularly sea-level rise, has added

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a new dimension to worldwide modifications of shorelines (Jones et al 2007a; Schlacher et al., 2008b). It has therefore become increasingly critical to understand how sandy beach ecosystems and the goods and services they provide will respond to this unprecedented environmental change.

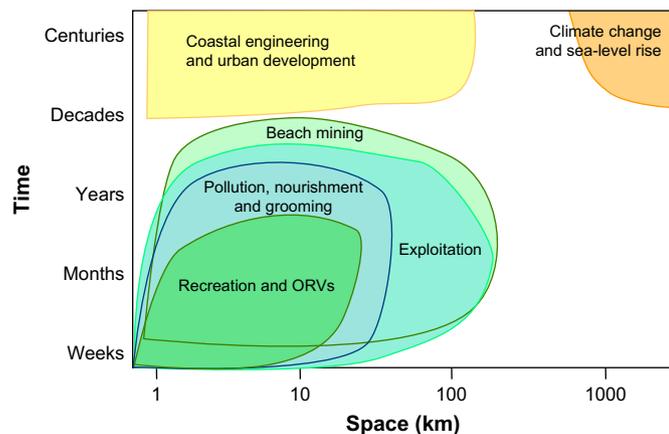
Exposed sandy beaches are physically dynamic habitats, inhabited by specialised biotic assemblages that are structured mainly by physical forces (reviewed in Defeo and McLachlan, 2005). Building on this perspective, documenting biotic responses to modifications of the physical environment is a critical step in predicting the consequences of global coastal change for beach ecosystems. Evidence for ecological change in beach ecosystems, which are exposed to human pressures at scales and intensities unmatched in history, is accumulating worldwide (Brown and McLachlan, 2002; Schlacher et al., 2007a).

Here we provide a brief synopsis of the unique physical and ecological attributes of sandy beach ecosystems and review the main anthropogenic pressures acting on the world's single largest type of open shoreline. Threats to beaches arise from a range of stressors that span a spectrum of impact scales from localised effects (e.g. trampling) to a truly global reach (e.g. sea-level rise). These pressures act at multiple temporal and spatial scales (Fig. 1), translating into ecological impacts that are manifested across several dimensions in time and space so that today almost every beach on every coastline is threatened by human activities.

## 2. Ecological features

### 2.1. The physical environment

Sandy beaches, defined by their sand, wave and tidal regimes, range from narrow and steep (reflective) to wide and flat (dissipative), as sand becomes finer and waves and tides larger; most beaches are intermediate between these extremes (Short, 1999; Finkl, 2004). Reflective beaches are coarse-grained and have no surf zones, whereas dissipative beaches have finer sediments and extensive surf zones. Filtration volumes are higher on



**Fig. 1.** Conceptual model and schematic diagram showing the relative spatio-temporal scales in which different impacts reviewed here generally operate on sandy beach macrofaunal communities. Boxes/envelopes indicate the potential extent of individual impacts in space and time with the lower curve reflecting the lower limit of impacts in time and space, whereas the upper curve reflects the corresponding maximum. Shorter term impacts (i.e., weeks to months) tend to be pulse disturbances and effects are generally expected to last for shorter time periods, since sandy-beach species are adapted to severe physical disturbances (e.g., storms). However, the temporal extent of impacts from the anthropogenic pressures depicted here could be drastically altered if the intensity of the disturbance is increased and/or its timing is more protracted. Under such circumstances, sandy beach habitats could become unsuitable for supporting macrofaunal communities and the ecosystem services they provide in the medium or long term.

permeable reflective beaches, mainly driven by wave action, and lower on dissipative beaches, where tidal action drives most water throughput. Thus filtration is rapid on reflective beaches and slower on dissipative beaches, but the sand body of all open beaches is well flushed and oxygenated; only under conditions of very fine sand, such as on some dissipative and low energy beaches, can reducing conditions develop in the deeper sediment layers (McLachlan and Turner, 1994).

Beaches are closely linked to surf zones and to coastal dunes through the storage, transport and exchange of sand; therefore impacts on beaches have consequences for these adjacent habitats (Komar, 1998). Sand transport, driven by waves on the wet side and wind on the dry side, is highest in exposed surf zones, whereas sand storage is often greatest in well-developed dunes. Sand tends to move rapidly seawards across the beach and surf zone during storms and to return more slowly landwards during calms. In this way, storm wave energy is dissipated and the soft coast is protected from permanent erosion (Short, 1999; Nordstrom, 2000).

### 2.2. Faunal components and ecosystem properties

The intertidal areas of beaches provide habitats for a diversity of fauna. The lacunar environment between the grains harbours interstitial organisms (bacteria, protozoans, microalgae and meiofauna), forming a distinct food web. Larger macrobenthic invertebrates burrow actively and include representatives of many phyla, but crustaceans, molluscs and polychaete worms are usually dominant and encompass predators, scavengers, filter- and deposit feeders. These macrobenthic invertebrates can reach high abundance (ca. 100,000 ind m<sup>-1</sup>) and biomass (>1000 g m<sup>-1</sup>), particularly in dissipative to intermediate beach types in temperate zones. Beaches that receive significant inputs of algae/seagrass wrack support a rich supralittoral fauna of crustaceans and insects. Most beach species are found in no other environment, their unique adaptations for life in these dynamic systems include: mobility, burrowing ability, protective exoskeletons, rhythmic behaviour, orientation mechanisms and behavioural plasticity (Chelazzi and Vannini, 1988; Scapini et al., 1995; Brown, 1996; Scapini, 2006).

The composition and abundance of invertebrate assemblages are controlled primarily by the physical environment, intertidal swash and sand conditions being harshest on reflective beaches and more benign on dissipative beaches. Consequently, more species can colonise dissipative beaches, but fewer species, mainly robust crustaceans, can establish populations on reflective beaches (McLachlan and Dorvlo, 2005). Whereas the effects of biological interactions (e.g. competition, predation) are overshadowed by physical factors on reflective beaches, they become more influential in structuring communities on dissipative beaches (Defeo and McLachlan, 2005).

Faunal patterns change on either side of the intertidal zone. Supralittoral zones are important nesting areas for turtles and shorebirds, and provide a favourable habitat for invertebrates on stable reflective beaches. The fauna of the lower beach may extend their distribution seawards into the turbulent surf zone, where zooplankton, shrimps and prawns can be abundant; surf zones are also important nursery and foraging areas for fishes.

Food webs of sandy beaches are mainly based on marine sources, such as phytoplankton, stranded algae, seagrasses and carrion (McLachlan and Brown, 2006). In dissipative systems, high productivity may be driven by surf phytoplankton and microorganisms, supporting benthic macrofauna and zooplankton as primary consumers and fishes and birds as top predators. While filtering water, the porous sand body and its biota mineralise organic matter and recycle nutrients, making beach ecosystems a crucial element in the nearshore processing of organic matter and nutrients.

Not only are beaches coupled to marine systems trophically, but they also interact physically and biologically with coastal dunes. Besides sediments, beaches and dunes exchange a variety of organic materials, and animals from both habitats move across the dune/beach interface to feed. Finally, beaches link terrestrial dune aquifers with coastal seas through the discharge of nutrient-rich groundwater.

### 2.3. Ecosystem services and sensitive features

Sandy shores provide a wide range of ecosystem services, many of which are essential to support human uses of sandy coasts. The most important ecosystem services include: (1) sediment storage and transport; (2) wave dissipation and associated buffering against extreme events (storms, tsunamis); (3) dynamic response to sea-level rise (within limits); (4) breakdown of organic materials and pollutants; (5) water filtration and purification; (6) nutrient mineralisation and recycling; (7) water storage in dune aquifers and groundwater discharge through beaches; (8) maintenance of biodiversity and genetic resources; (9) nursery areas for juvenile fishes; (10) nesting sites for turtles and shorebirds, and rookeries for pinnipeds; (11) prey resources for birds and terrestrial wildlife; (12) scenic vistas and recreational opportunities; (13) bait and food organisms; and (14) functional links between terrestrial and marine environments in the coastal zone.

Certain features and processes on sandy shores are particularly vulnerable to stressors or change: (1) large volumes of sand are transported under high-energy conditions, so the consequences of disturbing sand budgets are most severe in such situations; (2) under low-energy conditions, water circulation is limited and therefore dispersal of materials, such as pollutants, is slow; (3) foredune vegetation is easily disturbed with resultant destabilisation of dunes; (4) because many beaches rely on marine organic subsidies, removal of such inputs (e.g. beach grooming) has serious consequences for beach food webs; (5) vertebrates that nest on the backshore are considered especially important in coastal conservation, and both turtles and shorebirds are easily disturbed in their nesting areas.

## 3. Stressors

### 3.1. Recreation

Recreational seashore activities are overwhelmingly concentrated on sandy beaches. The effects of these pressures are particularly noticeable at scales ranging from weeks to months and from <1 to 10 km (Fig. 1). Burgeoning coastal populations, coupled with more leisure time and improved mobility, have escalated the intensity and spatial ambit of beach recreation over recent decades (De Ruyc et al., 1997; Caffyn and Jobbins, 2003; Fanini et al., 2006). Being the prime sites for human recreation, beaches underpin many coastal economies (Klein et al., 2004). Beach management therefore customarily focuses on maximising the recreational experience for beach users, which often results in ecologically harmful human interventions such as nourishment (Speybroeck et al., 2006), beach grooming (Llewellyn and Shackley, 1996; Dugan et al., 2003), coastal armouring (Dugan and Hubbard, 2006; Dugan et al., 2008), destruction of dunes to construct tourism infrastructure (Nordstrom, 2000), and light and sound pollution (Bird et al., 2004; Longcore and Rich, 2004).

Impacts caused directly by recreational activities are emerging as significant environmental issues (Schlacher et al., 2008b). Dune vegetation is vulnerable to mechanical impacts caused by trampling (Liddle and Grieg-Smith, 1975), and modern beach management practices progressively seek to restrict human access to these sensitive areas (Scapini, 2002). Nevertheless, camping and

driving activities, which severely impact dune vegetation, both continue unabated in many parts of the world (Luckenbach and Bury, 1983; Hockings and Twyford, 1997; Groom et al., 2007).

Evidence is sparse about how sensitive intertidal invertebrates might be to human trampling. Direct crushing of individual organisms on the unvegetated beach has been documented (Moffett et al., 1998), and macrobenthic populations and communities respond negatively to increased human activity levels (Weslawski et al., 2000; Fanini et al., 2005; Veloso et al., 2006), but not in all cases (Jaramillo et al., 1996). It can be difficult to separate the effect of human trampling from habitat modifications (e.g. seawalls replacing foredunes) because these often coincide in high-use areas (Barros, 2001).

Impacts of trampling on supralittoral fauna are usually seasonal at medium-high latitudes, enabling some recovery during periods of low use (see e.g., Scapini et al., 2005). However, press disturbances are expected on low-latitude beaches where the intensity of the disturbance is much higher than in temperate areas and its timing is more protracted. Under such circumstances, beach habitats could become unsuitable for sandy beach macrofauna in the medium or long term. It is therefore necessary to establish the ecological carrying capacity of beaches in terms of direct use, and to develop mechanisms for controlling access, tasks that many authorities neglect (De Ruyc et al., 1997).

Beaches and dunes are critical habitats for shorebirds and turtles (Hosier et al., 1981; Hubbard and Dugan, 2003) and both taxa are sensitive to disturbances. For example, human activities disturb shorebirds, modifying key behavioural traits that are crucial to their survival and reproduction (Burger, 1991, 1994; Lord et al., 2001; Verhulst et al., 2001), including: (1) changes to foraging behaviour resulting in less feeding time, shifts in feeding times and decreased food intake; (2) decreased parental care when disturbed birds spend less time attending the nest, thus increasing exposure and vulnerability of eggs and chicks to predators; and (3) decreased nesting densities in disturbed areas and population shifts to less impacted sites.

Off-road vehicles (ORVs) are commonly used on beaches and dunes worldwide (Godfrey and Godfrey, 1980; Priskin, 2003; Schlacher and Thompson, 2007) and cause damage that includes: (1) disturbing the physical attributes and stability of dunes and beaches by deeply rutting the sand surface and destroying embryonic foredunes in the tyre tracks (Anders and Leatherman, 1987; Kutiel et al., 1999; Priskin, 2003; Schlacher and Thompson, 2008); (2) destroying dune vegetation, leading to lower diversity and less floral ground cover (Luckenbach and Bury, 1983; Rickard et al., 1994; Groom et al., 2007); and (3) disturbing, injuring or killing beach fauna (Van der Merwe and Van der Merwe, 1991; Schlacher et al., 2007b, 2008a), including endangered vertebrates such as turtles and shorebirds (Hosier et al., 1981; Buick and Paton, 1989; Williams et al., 2004). Whether direct mortality of beach invertebrates caused by ORVs propagates to higher-order ecological effects, such as disruptions of food web linkages to fishes and raptors, is currently unresolved. Nevertheless, positive population responses by shorebirds following exclusion of ORVs (Williams et al., 2004) suggest that ORV-impacts can extend beyond the individual organism level of ecological organisation.

### 3.2. Cleaning

Cleaning or grooming, a common practice on beaches heavily used for tourism (Poinar, 1977; Llewellyn and Shackley, 1996; Engelhard and Withers, 1997; Dugan et al., 2003; Davenport and Davenport, 2006), clears beaches of macrophyte wrack, litter and other debris by raking and sieving the sand, often with heavy equipment (Kinzelman et al., 2003; Fanini et al., 2005). Grooming removes not only unwanted material, but also propagules of dune

plants and other species, and it perturbs resident organisms and roughens the sand, thereby exposing a greater surface area to the erosive effects of wind.

Wrack removal has significant ecological consequences, especially in regions with high levels of marine macrophyte production. The community structure of sandy beach macroinvertebrates can be closely linked to wrack deposits (e.g. Griffiths and Stenton-Dozey, 1981; Griffiths et al., 1983; Stenton-Dozey and Griffiths, 1983; McLachlan, 1985; McGwynne et al., 1988; Inglis, 1989; Colombini and Chelazzi, 2003; Dugan et al., 2003; de la Huz et al., 2005), which provide both a food source and a microhabitat refuge against desiccation (Colombini and Chelazzi, 2003). Wrack-associated fauna, such as talitrid amphipods, oniscoid isopods and insects, which can comprise up to 40% of the species and represent important prey for higher trophic levels (Dugan et al., 2003), are significantly reduced in species richness, abundance, and biomass by grooming (Dugan et al., 2003; Fanini et al., 2005). By contrast, taxa with well-developed dispersal abilities, such as flies, can be more prevalent on groomed beaches (Dugan et al., 2003).

Upper intertidal meiofauna may also be impacted by grooming, although these effects can be confounded with those from trampling (Gheskiere et al., 2005). Meiofauna communities may recover relatively quickly (24 h) from a single, short-term grooming event (Gheskiere et al., 2006), but it remains unknown what the consequences are of repeated, regular beach cleaning activities. Effects of this stressor could be noticeable at scales ranging from weeks to years and from <1 to 100 km (Fig. 1).

Because shorebird numbers are positively correlated with wrack cover and the biomass of their invertebrate prey that feed on wrack (Tarr and Tarr, 1987; Hubbard and Dugan, 2003; Dugan et al., 2003), grooming will lower bird numbers. The heavy equipment used in beach grooming can also cause direct mortality of the eggs and young of beach-nesting shorebirds, turtles and fish (Martin et al., 2006); many groomed beaches therefore no longer support breeding populations of these vertebrates. Furthermore, tracks created by grooming can disorientate turtle hatchlings trying to reach the sea (Hosier et al., 1981). Finally, grooming can result in abnormally broad unvegetated zones that are inhospitable to dune formation or plant colonisation, thereby enhancing the likelihood of erosion.

Invertebrates may recover slowly from the effects of grooming, especially if it is conducted on a daily or weekly basis throughout the year (Fig. 1), as is the case along many developed coastlines. For example, nearly half (>160 km) of the sandy beaches in southern California are regularly groomed (Dugan et al., 2003), and some beaches in Los Angeles are groomed twice a day (Dugan et al., 2000). Although beach cleaning is generally undesirable from an environmental conservation perspective, simple mitigation strategies could entail alternating areas of natural (uncleaned) beach with groomed sections, if grooming is deemed essential to provide clean beaches for recreation.

### 3.3. Nourishment

More than 70% of the world's beaches are experiencing erosion (Bird, 1996). Because engineering solutions, such as seawalls, breakwaters and groynes, are often ineffective to the point of causing the loss of the intertidal beach (Pilkey and Wright, 1989; Hsu et al., 2007), beach nourishment (also called beach replenishment, restoration or renourishment), has increasingly been used to combat shoreline erosion. Nourishment is preferred on both economic and conservation grounds (Finkl and Walker, 2004), but it can cause ecological damage to sandy beach habitats (Blott and Pye, 2004) and biota (reviewed in Goldberg 1988; Nelson, 1988; Peterson and Bishop, 2005; Speybroeck et al., 2006). The typical scale of nourishment is at 1–10 km (Peterson and Bishop, 2005) and

at temporal scales ranging from weeks to years (Fig. 1). Impacts occur both at sites from which sediment is extracted and at the receiving environment. These impacts manifest at the population (demography and dynamics), community (species richness) and ecosystem (functional processes, nutrient flux, trophic dynamics) levels. Affected biota include benthic micro-algae, vascular plants, terrestrial arthropods, intertidal and subtidal invertebrates, other marine zoobenthos and avifauna (Bishop et al., 2006; Peterson et al., 2006; Speybroeck et al., 2006; Fanini et al., 2007, 2009).

Factors influencing the nature and extent of ecological impacts of nourishment include the mechanical process itself, its timing, and the quality and quantity of new sediment placed (Speybroeck et al., 2006). Effects may be direct, such as mortality of organisms when buried, or indirect, such as reduced prey availability for shorebirds (Nelson 1993a, b; Bishop et al., 2006; Peterson et al., 2006). Most research has targeted the effects of altered sediment quantity and quality on macrofauna (Hayden and Dolan, 1974; Rakocinski et al., 1996; Peterson et al., 2000; Menn et al., 2003; Bilodeau and Bourgeois, 2004; Jones et al., 2007b). The immediate impacts are usually large and may be caused either by burial (Menn et al., 2003; Peterson et al., 2006; Jones et al., 2007b) or by emigration (Hayden and Dolan, 1974). These effects may be compounded by changes in beach morphology, particularly when nourishment creates a steeper beach and reduces the habitat area for some species (Peterson et al., 2006; Fanini et al., 2007, 2009). Nourishment can also disturb the nesting and foraging of birds, destroy dune vegetation and compact the sand (Speybroeck et al., 2006). Compaction affects the interstitial spaces, capillarity, water retention, permeability and the exchange of gases and nutrients.

Nourishment of sandy beaches usually acts as a short-term, pulse disturbance (Peterson and Bishop, 2005) that elicits a pulse ecological response (i.e. recovery occurs). This is expected since sandy-beach species are adapted to severe physical disturbances, storm events having been a frequent feature of their evolutionary history (Hall, 1994). Recovery probably occurs in months rather than years (Fig. 1), but the trajectory depends on sediment quality (Nelson, 1988, 1993a, b; Peterson et al., 2000, 2006). However, if the profile of the nourished beach and the imported sediments do not match the original conditions, recovery of the benthos is unlikely (Goldberg, 1988; Peterson et al., 2000, 2006). Thus, where unnaturally coarse or fine sediments are used, severe ecological impacts may occur and recovery is protracted (Rakocinski et al., 1996; Peterson et al., 2000, 2006). In addition, artificially flattened and extended sand bodies can be colonised by rapidly moving opportunistic macrofauna; under these conditions, few species dominate and biodiversity is reduced (Peterson and Bishop, 2005).

Mitigation of ecological impacts of nourishment is often impeded by limited data about the life history of the affected species, recovery rates and the cumulative effects of repeated nourishment events (Speybroeck et al., 2006). Nevertheless, basic management recommendations include: (1) the avoidance of sediment compaction; (2) careful timing of operations to minimise biotic impacts and enhance recovery; (3) the selection of locally-appropriate techniques; (4) the implementation of several small projects rather than a single large project, including repeated application of sediment in shallow layers (<30 cm) rather than in single pulses that kill the fauna by deep burial; (5) interspersing of nourished beach sections with unaffected areas; and (6) importing sediments and creating beach profiles that match the original beach conditions as closely as possible.

### 3.4. Pollution

Pollution is a sensitive issue on beaches given their immense value for recreation and tourism. Pollutants act at a variety of spatial and temporal scales (Fig. 1) and include a variety of

anthropogenic materials, ranging in size from molecules to large debris, and can impair the physiology, survival, reproduction and behaviour of species in all habitats of the beach from interstitial environments (McLachlan, 1977) to the surf zone (Noble et al., 2006). Pollution can also cause aesthetic disturbances, thereby impacting tourism industries that depend on public perceptions of clean beaches (Tudor and Williams, 2003).

Most solid waste stranded on the shore is allochthonous, brought ashore by waves and currents. Plastic is extremely persistent and it dominates the visible litter on sandy beaches worldwide (Derraik, 2002; Shiber, 2007). The major dangers associated with plastic include ingestion by and entanglement of vertebrates, such as seals, seabirds and turtles (Mascarenhas et al., 2004). In terms of human health, risk increases when medical plastic wastes reach the shore from coastal dumping sites (Sindermann, 1996). Economic losses can arise when tourist beaches are persistently contaminated by litter from land or ocean sources.

Wastewater and sewage are significant sources of beach pollution. Intertidal sands (Ulfig et al., 1997; Salvo and Fabiano, 2007) and surf-zone waters (Bonilla et al., 2006, 2007; Noble et al., 2006) can be contaminated by pathogens, including bacteria and filamentous fungi, which are delivered to the sea by sewage systems discharging directly into coastal waters or estuaries near beaches (Mardon and Stretch, 2004; Araujo and Costa, 2007). Bacterial levels in surf zone waters that exceed human health standards are a frequent cause of beach closures and public warnings on many developed coasts. Contamination at any given beach is dependent on: (a) the appropriateness of waste-water management practices; (b) the timing and intensity of local rainfall events and subsequent runoff; and (c) the strength of mixing and dispersion in the surf zone (Stretch and Mardon, 2005). Metal pollution from wastewater and industry accumulates preferentially on fine-grained beaches (Ramirez et al., 2005), which usually support relatively high diversity and biomass, thus causing significant reductions both in biodiversity and in the population density of economically valuable species (Castilla, 1983; Haynes et al., 1997).

Freshwater effluents arising from human activities can also cause a broader deterioration in the quality of the surrounding habitat (Lercari et al., 2002), with impacts at various levels of ecological organisation, ranging from individuals and populations, which might experience reduced survival, growth and fecundity rates (Defeo and de Alava, 1995; Lercari and Defeo, 1999; Lozoya and Defeo, 2006), through to the community, which can exhibit changes in diversity and structure (Lercari et al., 2002; Lercari and Defeo, 2003; Defeo and Lercari, 2004).

Oil spills are potentially the most destructive pollution source impacting sandy beaches, affecting all trophic levels (Bodin, 1988; Suderman and Thistle, 2003). Impacts can be acute and temporary, but they can also be more chronic, lasting for many months or even years (Irvine et al., 2006). Beach morphodynamics and exposure strongly influence the duration of contamination: the coarser the sediment (i.e., reflective beaches), the more rapidly and deeply oil penetrates, sometimes even reaching below the groundwater table (Bernabeu et al., 2006). Sheltered beaches are generally more sensitive to pollution than exposed beaches, even when sediments are fine and oil does not penetrate deeply, because they are less well flushed by wave action and subsurface oil persists much longer than surface oil (Sindermann, 1996). Persistence and breakdown of stranded oil depends on sand particle size, wave energy, temperature and other factors (Owens et al., 2008), including fungal degradation (Elshafie et al., 2007). Methods used for oil spill cleanup on beaches may also cause ecological impacts. Public concern about oil spills has focused on catastrophes like the IXTOC 1 in Mexico (Rabalais and Flint, 1983), the Exxon Valdez in Alaska (Short et al., 2004) and the Prestige in Spain (de la Huz et al., 2005)

and many smaller events tend to go unreported, but introduce considerable quantities of oil into the beach environment. Tar balls that originate from the (often illegal) flushing of ships' bilges at sea are present on most beaches, especially near oil shipping routes (Corbin et al., 1993; Coles and Al-Riyami, 1996; Abu-Hilal and Khordagui, 2007).

In most cases, management measures against pollution have to be taken without a preceding environmental impact assessment, as has been the case for oil spills affecting beaches all over the world. Oil pollution impacts differ between chronic cases and episodic events. The former seldom require direct action, whereas the latter may need a major cleanup, which in turn causes impacts. Management requires monitoring and control of water and sediment quality, particularly in urban areas (Pereira et al., 2003).

### 3.5. Exploitation

Artisanal invertebrate fisheries are the most common form of exploitation on sandy beaches (McLachlan et al., 1996; Kyle et al., 1997). Although these are generally of reasonably small scale, impacts can be significant (Fig. 1). First, target species tend to occur in patches (Defeo and Rueda, 2002; Schoeman and Richardson, 2002), and fishers can therefore easily target and serially deplete dense aggregations (Caddy and Defeo, 2003; Pérez and Chávez, 2004). Most exploited sandy beach stocks are also short lived and susceptible to recruitment overfishing in the short term (Defeo, 1996b). Moreover, harvesting activities cause incidental mortalities, both directly through physical damage of organisms and indirectly when sediment disturbance lowers habitat quality and suitability (Sims, 1997; Defeo, 1998). Such incidental mortality is not limited to the exploited fraction of the target populations, but also affects their unexploited fractions and non-target species on fished beaches (Defeo 1996a, 1998).

Long-term, large-scale field experiments have demonstrated a predictable pattern of responses to fishing (Defeo, 1996a, 1998; Castilla and Defeo, 2001): elevated harvesting rates significantly reduce recruitment to the exploitable stock and to the population as a whole, resulting in a declining spawning stock, a decreasing age/size at maturity and a decreasing proportion of older individuals in the catch. Several fishery-related density-dependent processes have also been identified. For example, fishery closures can result in elevated adult densities, which cause density-dependent increases in natural mortality rates of older clams (Defeo, 1996a), as well as decreases in growth and survival rates of young-of-the-year (Defeo, 1998), the inhibition of recruitment of both target and sympatric species, and the reduction of age-specific fertility (Defeo and de Alava, 1995; Brazeiro and Defeo, 1999; Lima et al., 2000).

Where fishing-related disturbance is a pulse of limited spatial extent, its impacts can be ameliorated relatively quickly (Schoeman et al., 2000). However, chronic disturbance can cause large-scale and persistent impacts for populations and communities, as documented for Argentinean beaches where fishing involving the use of tractors caused lasting damage to populations of yellow clam *Mesodesma mactroides* for decades (Defeo, 2003). Detrimental effects of fishing can also be exacerbated when they are superimposed on environmental change, irrespective of whether such change is of anthropogenic origin, or not (Defeo, 2003). On a New Zealand beach, clam populations depleted by artisanal fisheries between the mid-1960s and 1990 failed to recover following the closure of the fishery (McLachlan et al., 1996), apparently because continuous erosion of beaches has dramatically reduced the clam habitat (Beentjes et al., 2006). Similarly, the effects of fishing can be amplified or reinforced by mass mortalities associated with blooms of toxic algae, by parasitism and by temperature anomalies (Arntz et al., 1987; Defeo, 2003; Fiori et al., 2004).

Technological interdependencies among user groups and ecological interdependencies between stocks are common in sandy beach fisheries (Defeo, 2003). Even fisheries in which juveniles and adults are spatially segregated are not immune to this phenomenon, with separate user groups often targeting intertidal and subtidal stocks using different gear types (McLachlan et al., 1996). Moreover, commercial fishers and recreational anglers, who collect large specimens for bait, tend to be more selective than recreational food collectors (Schoeman, 1996; Murray-Jones and Steffe, 2000). In this way, different user groups, which are often difficult to distinguish, can inflict different patterns of fishing mortality on the resource. Thus, beach fisheries tend to involve a complex blend of users, requiring the explicit incorporation of cultural, social and political dimensions in resource management (Castilla and Defeo, 2001). As harvesting not only affects the targeted species directly through fishing mortality, but it also has collateral impacts that can alter ecosystem structure and functioning, sandy beach fisheries can have impacts disproportionate to their economic value. Being vulnerable to fishing, few beach fisheries have scope for growth. Because it is difficult to restrict participation in these accessible fisheries, management failure and stock depletion seem to be the norm rather than the exception (reviewed in McLachlan et al., 1996; Defeo, 2003).

### 3.6. Biological invasions

Human activities that are vectors for species introductions to beaches are not a recent phenomenon. Among the oldest-known human agents of species introduction to beaches is the practice of exchanging large quantities of “dry” ballast sand and gravel on shore during ship maintenance, which dates back to the early days of ocean-going sailing vessels (Carlton, 1989).

Nearshore and intertidal benthic habitats in many regions have been invaded by alien macrophytes that displace native algae and seagrasses (Russell, 1992; Boudouresque et al., 1995; Inderjit et al., 2006). Although not invaders of beaches in the sense of establishing there, they can form large wrack deposits on sandy beaches when detached from their benthic habitats. In Hawaii, accumulations of the invasive *Hypnea musciformis* of >9000 kg week<sup>-1</sup> have been reported for pocket beaches. In Argentine Patagonia, the biomass and composition of macrophyte wrack on beaches have been changed by the invasive kelp *Undaria pinnatifida* (Piriz et al., 2003). The kelp *Sargassum mutica* is accumulating in large quantities on beaches of northwestern Spain (Lastra, personal observation), and the green algae *Caulerpa taxifolia* can replace the native seagrass *Posidonia oceanica* as the main component of wrack on Mediterranean beaches (Boudouresque et al., 1995; de Villele and Verlaque, 1995). The preference of wrack consumers (e.g., talitrid amphipods) for macroalgae suggests the potential for impacts of invasive algae on food webs and nutrient cycling on beaches (Lastra et al., 2008).

Few invasive invertebrates have been reported from sandy beaches to date, but this may reflect poor sampling coverage rather than a lack of biological invasions. The semi-terrestrial talitrid amphipod *Orchestia cavimana*, originally from freshwater habitats in the Black Sea and Mediterranean region, has been reported from fresh to brackish habitats in the British Isles, and more recently has been spreading rapidly on open exposed shores with low salinities of the NE Baltic Sea (Herkul et al., 2006). The talitrid *Platorchestia platensis*, which presently inhabits Europe, the Canaries, Japan, India, Hawaii and other Pacific islands, and which was recently reported in the Baltic, may be an exotic species (Spicer and Janas, 2006). A few widespread polychaetes (e.g., glycerids and spionids) reported from exposed beaches, are classified as cryptogenic, implying that their origin is unknown and that they are neither demonstrably native nor introduced (Carlton, 1996). The best known of these are *Glycera americana* and *Scololepis squamata*, both

of which are reported from beaches and estuaries of several continents (McLachlan and Brown, 2006). The possibility that some cryptogenic forms could represent cryptic species rather than a single widespread invasive species needs to be considered.

Invasive terrestrial insects can also impact beach ecosystems. Red fire ants (*Solenopsis invicta*) from Brazil, are now found in the USA, Australia, Taiwan and China; they can prey on the eggs and hatchlings of loggerhead and green sea turtles (Allen et al., 2001). A congener, the black fire ant (*Solenopsis richteri*) preys intensely on intertidal polychaetes in Argentinean lagoons (Palomo et al., 2003), which suggests that it could also consume intertidal invertebrates on exposed beaches. The Argentine ant (*Linepithema humile*, formerly *Iridomyrmex humilis*), a small, dark ant native to northern Argentina, Uruguay, Paraguay, and southern Brazil, has invaded a variety of habitats, including beaches, in South Africa, New Zealand, Japan, Australia, Europe and the USA.

### 3.7. Coastal development and engineering

The expanding use of coasts by human populations characteristically involves a gradual intensification of urban development in the littoral-active zone, with the ultimate consequence that coupled surf-beach-dune systems must be managed (Nordstrom, 2000). This management most commonly focuses on the sediment budget. Modern coastlines are increasingly starved of sand as dams trap sediments that would otherwise feed beaches; the sediment budget is further disrupted by activities such as quarrying, land reclamation, urbanisation, afforestation and agricultural use (Nordstrom, 2000; Sherman et al., 2002). As a result, most modern coastlines are experiencing accelerating rates of erosion (Innocenti and Pranzini, 1993; Cooper and McKenna, 2008).

Society's response to beach erosion and shoreline retreat relies heavily on engineering interventions that place armouring structures on beaches (Nordstrom 2000, Charlier et al., 2005, Griggs, 2005a, b). Hard structures, such as walls constructed of stone, concrete, wood, steel or geotextiles, have been used for centuries as a coastal defence strategy (Charlier et al., 2005), but this protection is not achieved without ecological costs. Armouring structures alter the natural hydrodynamic system of waves and currents, thereby affecting sand transport rates, which in turn control the erosion-accretion dynamics of beaches (Miles et al., 2001; Hsu et al., 2007). Intertidal seawalls and other structures that reflect wave energy and constrain the natural landward migration of the shoreline have unplanned environmental impacts, such as flanking erosion of shorelines adjacent to those protected by engineering structures (Hall and Pilkey, 1991; Weigel 2002a, b, c; Griggs, 2005a, b). Passive erosion occurs on the armoured beaches themselves; and because seawalls arrest the landward retreat of the shoreline in the face of erosion, the beach seawards of the structure is frequently drowned (Hall and Pilkey, 1991; Fletcher et al., 1997; Griggs, 2005b). Armouring might also enhance beach erosion on protected coasts, although this remains controversial (Kraus and McDougal, 1996; Griggs, 2005b).

The impacts of seawalls and other coastal armouring structures may cause significant habitat changes, with attendant ecological impacts (Sobocinski, 2003; Martin et al., 2005; Dugan and Hubbard, 2006; Bertasi et al., 2007) that can be difficult to detect in the short term (Jaramillo et al., 2002). As eroding beaches become narrower after armouring, the reduced habitat can directly lower the diversity and abundance of biota, especially in the upper intertidal zone (Sobocinski, 2003; Dugan and Hubbard, 2006; Dugan et al., 2008). This, in turn, can also be detrimental to higher trophic levels, e.g., coastal avifauna may be impacted both by reduced habitat area and by declining intertidal prey resources. This phenomenon is reflected by observations of significantly lower numbers and fewer species of birds on armoured compared with unarmoured

segments of Californian beaches (Dugan and Hubbard, 2006; Dugan et al., 2008). The loss of dry-sand following armouring also eliminates nesting habitats for turtles and some specialised species of fishes (e.g. California grunion, surf smelt in the USA).

Armouring may also affect the quality of beach habitats. For example, the rates of deposition and retention of macrophyte wrack, driftwood and other natural debris that can be important to beach biota as food and habitat, are lower on armoured than unarmoured beaches (Sobocinski, 2003; Dugan and Hubbard, 2006). By contrast, increased deposition of wrack can occur on some beaches where offshore structures have been placed (Martin et al., 2005). These offshore defences also alter physical characteristics and benthic communities of beaches and surf zones (Martin et al., 2005; Bertasi et al., 2007).

Because shore-protection structures often present novel habitats on beaches, they could promote the introduction of exotic species. Together, engineering structures and introduced species could severely impact native assemblages, not only by modifying selection pressures as a result of habitat transformation, but also by introducing competition (Gonzalez et al. 2008). However, this issue remains speculative and more work is needed to assess its importance for sandy beach ecosystems.

Sea-level rise and other large-scale and long-lasting effects of global warming (Fig. 1) are expected to intensify beach erosion (Slott et al., 2006). To protect human assets on soft-sediment coastlines globally in the face of escalating erosion, the use of coastal armouring will therefore expand substantially. For extensively armoured coastlines in many countries (Fletcher et al., 1997; Griggs, 1998; Nordstrom, 2000), changes in sea level will shift the locations of existing armouring structures to lower positions on the shore, amplifying interactions with waves and tides and further accelerating beach erosion (Cooper and McKenna, 2008). Thus, the combined effects of rising sea levels and coastal armouring are predicted to cause ecological impacts to beaches on unprecedented scales.

### 3.8. Mining

Beach sands have been mined in many places, including Sri Lanka for building lime from coral sand (Clark, 1996), Tanzania, Korea, Argentina, Italy and many other countries for building sand (Masalu, 2002; Cho, 2006; Pousa et al., 2007), the Namibian coast for diamonds (McLachlan, 1996; Theron et al., 2003), and several areas for heavy minerals such as titanium and zirconium (Mulaba-Bafubandi et al., 2002; Panigrahi, 2005; Pirkle et al., 2005; Ghosh et al., 2006). Further, mine tailings are discharged onto beaches in many places. Diamond mining activity in Namibia has been shown to negatively impact shorebirds (Simmons, 2005), and tailings from copper mines cause lower meiofauna density and diversity on Chilean beaches (Lee and Correa, 2005). Removal of sand disturbs sediment budgets, possibly contributing to enhanced erosion (Masalu, 2002; Thornton et al., 2006); it may also alter particle size, changing the morphodynamic state of beaches and intertidal benthic communities (McLachlan 1996). Thus, the general effects of any form of mining on beaches are to damage the beach and dune habitat, to alter the sediment budget and hasten erosion. It is now illegal in many countries to remove sand from the littoral active zone and the practice should in general be prohibited (Clark, 1996). In many places sand is mined offshore, often on the outer shelf; the ecological impacts of this activity are probably slight for beaches, but not well understood (Byrnes et al., 2004).

### 3.9. Climate change

Although the exact magnitude of physical changes resulting from global climate change is still uncertain (IPCC, 2007), ecological

responses (e.g., changes in phenology, physiology, range and distribution, assemblage composition, species interactions) are increasingly apparent in sandy beaches (Brown and McLachlan, 2002; Jones et al., 2007a). However, there are no direct studies of effects of this long-lasting and large-scale stressor (Fig. 1) on beach ecosystems; consequently, many of our predictions of the likely impacts on beach ecosystems are derived from other systems.

Rising temperatures may have different implications at different latitudes and for taxa with different dispersive abilities and ranges. Because many sandy beach species (e.g. peracarid crustaceans) lack dispersive larval stages, their rates of range extension might be outpaced by changes in temperature, making these taxa particularly vulnerable to climate-change effects. Narrow-range endemic species would be at greatest risk (O'Hara, 2002) and some species could eventually be replaced by species from lower latitudes. But even migratory species, as well as those that have pelagic larval stages, may be impacted directly by large-scale, climate-driven changes to prevailing oceanographic systems.

Beach biota may also respond to indirect effects of temperature change on beaches. For example, comparatively small increases ( $\sim 0.6^\circ\text{C}$ ) in temperature have been associated with major changes in planktonic ecosystems in the North Atlantic over the past five decades (Richardson and Schoeman, 2004). Given that plankton is a key food source for suspension-feeding beach species, changes in plankton communities will have unpredictable impacts on sandy beach macrofauna. Changes may be significant for semi-terrestrial species (peracarids and insects), which will be affected both directly and indirectly by changes in water and air temperatures.

A direct consequence of warming seas is sea-level rise. Current estimates suggest that sea level is rising by an average of  $1.7 \pm 0.5 \text{ mm yr}^{-1}$  (IPCC, 2007), but there is potential for far greater rates of change (Rahmstorf et al., 2007). Irrespective, rising sea level pushes the high-water mark landward, causing beaches to migrate slowly inland. Low-gradient dissipative shores, which house the greatest biodiversity, are at most risk due to their erosive nature and the much greater run-up of swashes on gentle gradients. Moreover, warmer air and sea temperatures would translate into more frequent and more severe storms (IPCC, 2007), thus escalating beach erosion to the point where entire beaches could disappear, removing habitat for the biota. The introduction of setbacks is currently international best practice. This involves defining a shore-normal zone within which new development is prohibited and space is thereby provided for natural retreat of the shore. Complementary to this is managed coastal retreat, which involves preparing to abandon existing infrastructure within the setback zone and/or developing plans to remove these existing structures that are too close to the shore. Soft solutions, such as nourishment, should be sought first, with hard engineering the very last resort.

Predicted changes in ocean acidification could further impact sandy beaches. The continued reduction of pH in surface waters (0.1 unit less than pre-industrial levels and predicted to decline by a further 0.3–0.4 units by the end of the century) will reduce calcification rates and calcium metabolism in marine organisms (Feely et al., 2004), including several sandy beach molluscs and crustaceans (Hall-Spencer et al., 2008).

Management to maintain beach habitat requires long-term mitigation and/or adaptation strategies. Mitigation would arrest climate change and its consequences via large reductions in greenhouse emissions. Adaptation would provide setback zones, allowing the beach to migrate inland as the sea rises. The latter would have minimal ecological consequences for beaches, but would be very expensive in urban areas. Alternatively, if engineering solutions (e.g. seawalls) are used to defend societal assets, intertidal sand habitat will be lost (Finkl and Walker, 2004). Adaptive measures accept the reality of sea-level rise and coastline retreat and seek to increase coastal resilience, a concept with

ecological, morphological and socio-economic components (Carpenter and Folke, 2006). Unfortunately, coastal resilience has been impaired by anthropogenic effects on the sediment budgets of beach-dune systems (Alonso and Cabrera, 2002; Sherman et al., 2002; Tomlinson, 2002). Measures to promote resilience include the protection, vegetation and stabilisation of dunes, the maintenance of sediment supply and the provision of buffer zones, rolling easements or setbacks that allow the landward migration of the coastline.

#### 4. Discussion and conclusions

The major long-term threat facing sandy beaches worldwide is coastal squeeze, which leaves beaches trapped between erosion and rising sea level on the wet side and encroaching development from expanding human populations on land, thus leaving no space for normal sediment dynamics. Because of the inertia associated with global climate change and human population growth, no realistic management scenario will alleviate this threat in the short term. Instead, management must be based on existing knowledge, suggesting that the immediate priority is to avoid further development of coastal areas likely to be directly impacted by retreating shorelines. Under climate change, this zone must be shifted inland to create substantial setbacks that can account for the predicted inland movement of coastlines over the next decades. For example, pending legislation in KwaZulu-Natal (South Africa) formalises setback lines at the 10-m elevation contour, which should account not only for several decades of sea-level rise, but also for rates of erosion that are disproportionately faster than sea-level rise. Similar legislation exists elsewhere, and setbacks are widely recognised as critical (Healy, 2002; Komar et al., 2002; Daniel and Abkowitz, 2005; Jackson, 2005).

In the longer term, robust and efficacious management interventions must be founded in a more comprehensive ecological understanding of beaches. In this context, the spatial and temporal scales used to evaluate impacts on sandy beaches are particularly relevant. The scales of impacts to beach ecosystems (Fig. 1) encompass multiple orders of magnitude in both time and space, and include pulse and press drivers. Press disturbances (whatever the source of impact) are becoming increasingly common; they operate on time scales of years to decades and they may persist at least as long as the impacts remain. Our summary of the estimated spatio-temporal scales of human impacts on sandy beaches indicates that many of the impacts manifest at the scale of tens to hundreds of kilometres and last for months to years (Fig. 1). The cumulative effects of these impacts may already extend to regional and global scales for developed and urbanised coasts, as well as coasts used intensively for recreation (Cooper and McKenna, 2008; Halpern et al., 2008). However, long-term data sets describing either the natural dynamics of beach systems or the magnitude of human impacts on beaches are scarce and fragmentary. Thus, the necessary research priority is to implement long-term monitoring programmes that quantify the dynamics of key ecological attributes on sandy beaches at different organisational levels ranging from individuals to ecosystems, as well as to compile regional and global databases of empirical measurements of ecosystem condition. This should be useful to detect regime shifts and their corresponding drivers, which have been increasingly documented for some terrestrial, freshwater and marine systems (de Young et al., 2008), but not for sandy beaches yet.

As the number and magnitude of global change drivers increases over time, scaling up the findings of sandy beach research to larger and longer scales is a short-term need. To this end, multi-year, large-scale data sets should be built in order to assess the extent of range contractions or expansions and even the potential extinction of some species. Given that critical stressors faced by

sandy beach ecosystems, such as rising sea level and intensification of onshore winds, can overwhelm local patterns and processes, the interplay between fine-scale (e.g., erosion and local variations in abundance and other population descriptors) and broad-scale (e.g., macroscale variations in temperature and variations in distribution ranges) phenomena (i.e., a cross-scale interaction approach sensu Peters et al., 2008), is needed to provide a better understanding of the dynamics of sandy beach ecosystems. This should be operationalised through the creation of networks of long-term observation sites that should be interdisciplinary in nature (Carpenter, 2008 and references therein). Cooperative interdisciplinary investigation is thus a pressing need to assess the effects of human impacts on sandy beach ecosystems.

There is also scope for improvement in experimental design to better distinguish natural variability from anthropogenic impacts (Peterson and Bishop, 2005). It can be extremely difficult to separate the individual effects of different impact sources (e.g., human trampling vs. seawalls replacing foredunes), or to find unaffected beaches that could act as truly independent controls in experiments. In these situations, field experiments could be improved by interspersing sites across areas of beach with as full a spectrum of impacts as is possible. Because good control sites are very difficult to find (however, see Peterson et al., 2006 and Schlacher et al., 2007b for illustrative examples on sandy beaches), distinguishing natural variability from anthropogenic impacts can only be attempted by modelling gradients away from the impact sources, as shown by long-term (Defeo and de Alava, 1995) and large-scale (Lercari and Defeo, 1999, 2003) field studies. Therefore, the unambiguous definition of the relevant scales of analysis, which will vary according to the nature and extent of the impact (Fig. 1) and the organisational level under analysis, will be critical in developing a robust hypothesis testing framework.

Exposed sandy beaches have fewer invasive species than other coastal habitats (e.g. Wasson et al., 2005). This raises the questions of whether: (1) exposed beaches are more difficult to invade and colonise; and (2) the high frequency and intensity of disturbance on ocean beaches impede the establishment and survival of invasive species. In this way, sandy beaches may serve as a useful model system in which to test general ecological theories about the relationship between physical disturbance regimes and invasion success by alien species.

Because it is difficult to completely exclude human activities from sensitive coastal areas, zoning strategies will be important in measuring and managing spatial gradients of human impact. In this sense, areas of relatively natural, unaffected beach could be interspersed with areas where recreational activities are permitted. Here, the application of strategic management (Micallef and Williams, 2002), systematic conservation planning (SCP, Margules and Pressey, 2000) and marine spatial planning (MSP, Ehler, 2008) approaches could assist beach managers in determining which spatially explicit elements should receive conservation priority. SCP and MSP could be used to determine the optimal along-shore distribution of human activities, on the basis of knowledge about the current distribution of ecological values and human threats, coupled with an understanding of how these might change through time and which tools are available to manage them.

Marine reserves (MRs) and marine protected areas (MPAs) have not been widely implemented in sandy beaches. MRs could be a key tool for biodiversity conservation and should also facilitate spill-over effects into adjacent beach habitats. Given that most invertebrates of beaches are short-lived, the efficacy of MRs in conserving sandy beach ecosystems may be detectable quite rapidly. Design and allocation of MR networks should therefore become a top conservation priority for sandy beach ecosystems globally. Coastal networks combining MRs, MPAs with limited level of human activities, co-management and exploitation areas sustaining

exclusive community rights, could represent a short- to mid-term management objective that balances bio-socio-economic factors (Defeo and Castilla, 2005). Thus, management policies for beaches should integrate the interacting natural, socio-cultural and management systems to protect biological diversity and maintain essential ecological processes and life-support systems (James, 2000; Ariza et al., 2008).

A common feature of setback and zoning strategies is that they need to be enforced through legislation. Thus, all relevant stakeholders should be included from the beginning in the design, implementation and institutionalisation of spatial planning initiatives (Plasman, 2008). New perspectives for rational management of sandy beaches require paradigm shifts, including not only basic ecosystem principles, but also incentives for effective governance and sharing of management roles between government and local stakeholders (Caffyn and Jobbins, 2003; Castilla and Defeo, 2005). Legitimising the participation of stakeholders in the planning and surveillance of management measures is one promising short-term solution to current problems faced in sandy beach ecosystems, promoting compliance with regulations.

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