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Impact of urbanization on coastal wetland structure and function

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Abstract Urbanization is a major cause of loss of coastal wetlands. Urbanization also exerts significant influences on the structure and function of coastal wetlands, mainly through modifying the hydrological and sedimentation regimes, and the dynamics of nutrients and chemical pollutants. Natural coastal wetlands are characterized by a hydrological regime comprising concentrated flow to estuarine and coastal areas during flood events, and diffused discharge into groundwater and waterways during the non-flood periods. Urbanization, through increasing the amount of impervious areas in the catchment, results in a replacement of this regime by concentrating rain run-off. Quality of run-off is also modified in urban areas, as loadings of sediment, nutrients and pollutants are increased in urban areas. While the effects of such modifications on the biota and the physical environment have been relatively well studied, there is to date little information on their impact at the ecosystem level. Methodological issues, such as a lack of sufficient replication at the whole-habitat level, the lack of suitable indices of urbanization and tools for assessing hydrological connectivity, have to be overcome to allow the effects of urbanization to be assessed at the ecosystem level. A functional model is presented to demonstrate the impact of urbanization on coastal wetland structure and function.

Key words: coastal wetland, hydrological regime, nutrient, pollutant, sedimentation, urbanization

ECOSYSTEM SERVICES PROVIDED BY COASTAL WETLANDS

Wetlands are biologically diverse and productive transitional areas ('ecotones') between land and water, characterized by shallow water overlying waterlogged soil and interspersed submerged or emergent vegetation. By occupying zones of transition between terrestrial and marine ecosystems, coastal wetlands, including *Melaleuca* swamps, salt marshes, mangroves, intertidal mudflats, seagrass beds and shallow subtidal habitats, are the interface of the coastal landscape. It is generally understood that wetland formations are determined by the cumulative interactions of hydrology, landscape position, sediment dynamics, storm-driven processes, sea level rise, subsidence, and colonization and disturbance by animals (Varnell *et al.* 2003).

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Coastal wetlands have been suggested to offer many important ecosystem services (Woodward & Wui 2001), but few of these have been demonstrated, including the highly controversial 'outwelling' role (Odum 1980), which hypothesizes that coastal wetlands export large percentages of their production to support offshore secondary production (Lee 1995). Being productive and often spatially diverse habitats, wetlands fulfil important functions such as providing habitats for flora and fauna including migratory birds (Pethick 1984; Nakamura *et al.* 1997; Messina & Connor 1998; Stumpf & Haines 1998; Dierschke *et al.* 1999; A.P.M.W.C.C. 2001; Ishikawa *et al.* 2003) and help to moderate water quality (Faulkner 2004). Other migratory species, for example, fish, turtles and cetaceans also utilize inshore wetlands. Coastal wetlands such as mangroves, salt marshes, intertidal mudflats and seagrass beds produce a large variety of food to consumers, with system net primary productivity levels of mangroves and salt marshes often exceeding 2000 g and in some cases reaching 4000 g dry wt m^{-2} year^{-1} (Alongi 1998). In addition to providing habitat and food sources for associated organisms, wetlands may also support commercial and recreational fisheries through their role as nursery habitats (Lee 1999; Clynick & Chapman 2002; Ansari *et al.* 2003) and deliver several direct and indirect services to the local population (Bird 1984). Wetlands may also act as a buffer between land and sea as they prevent erosion, reduce currents, attenuate waves and encourage sediment deposition and accretion (Bird 1984). Analysis of the recent Asian tsunami suggests that there may be an inverse relationship between mangrove presence and tsunami damage (e.g. Dahdouh-Guebas *et al.* 2005).

Safeguarding such vital roles requires resource management strategies that are often at odds with increasing human development pressures (Stumpf & Haines 1998). In contrast to the high ecological value placed on coastal wetlands, they are often utilized for a number of destructive and consumptive uses such as waste dumping, land reclamation, aquaculture ponds, and dredging for navigational channels and marinas. Such activities have resulted in the recent rapid loss of coastal wetland habitats, such as salt marshes and mangroves. For example, >50% of mangrove forests present in many south-east Asian countries in the 1960s have already been lost to some form of development (Wilkinson *et al.* 1994), and the trend is continuing (Ewel *et al.* in review). Ironically, such destruction is concomitant to increasing awareness of the ecosystem services provided by coastal wetlands. One important emerging concept is that coastal wetlands deliver beneficial ecosystem services, such as supporting fishery production, as a complement of habitats rather than in isolation (e.g. Lee 2004), which has strong implications for their conservation and management. In a meta-analysis, Woodward and Wui (2001) demonstrated that a site-specific approach may be necessary in evaluating wetland services. Coastal wetlands also provide significant non-consumptive services, such as tourism and conservation of biodiversity resources, but the value of these services is often difficult to quantify.

URBANIZATION IMPACTS ON COASTAL WETLANDS

Urbanization is often referred to as either the degree of or increase in urban character or nature, and may either refer to a geographical area combining urban and rural areas, or refer to the transformation of areas to greater urban development. The term may be used to describe a condition at a specific time, namely the proportion of total population or area in urban localities or regions, or the increase of this proportion over time.

Globally, wetland ecosystems are under pressure from rapidly increasing urban populations in coastal areas (Ehrenfeld 2000). For example, coastal counties within 80 km of the coast make up only 13% of the land area of the continental USA, but this area encompasses 51% of the population (Rappaport & Sachs 2003). Within Australia, approximately 84% of the population live within the coastal region (Australian Bureau of Statistics 2002). Outside Australian capital cities, most population growth has occurred in coastal areas such as the city of Gold Coast in Queensland, which increased by 240 500 people between 1986 and 2003 (Australian Bureau of Statistics 2002; Office of Urban Management 2004). The continuing population growth within coastal regions ensures that there will be ongoing impacts on coastal wetland ecosystems (Callaway & Zedler 2004).

Hydrological and sedimentation regimes are the main physical drivers in coastal wetlands. Both are often significantly modified by human activities and introduce additional drivers such as pollutants, exotic species, harvesting of biomass and direct habitat loss. The impact of urbanization on coastal wetlands is initially and mainly through alteration of hydrological and geomorphological processes (Table 1). Structural and functional ecological changes then follow. The intensities of these disturbances vary along different spatial and temporal scales (Lindegarth & Hiskin 2001). Impacts caused by one type may be modified by other disturbances within the system, creating complex interactions between them. For example, as a result of urbanization led changes in sediment loads, water quality, aquatic systems and associated organisms within affected areas may be exposed to elevated concentrations in trace metals, pesticides, hydrocarbons and nutrients (Horner 2000; Burton *et al.* 2004). Additionally, other types of disturbances in areas affected by urban development may include: structural and hydrological modifications (including altered storm water run-off, drainage and filling characteristics), ecological modifications, recreational activities; the introduction of exotic species and resulting effects from increased sedimentation loads (Ehrenfeld 2000; Horner 2000; Lindegarth & Hiskin 2001). Other disruptions include those to wildlife migration routes.

This review will focus on the general impacts of urbanization on coastal wetland structure and function, through altering the physical environment such as the hydrological regime and sediment dynamics; and structure of the biota and prevalent trophodynamics. In view of the confusion over the definition of coastal wetlands, the scope of this review is limited to intertidal and shallow (<6 m at low tide) subtidal soft-sediment habitats, with particular reference to estuarine wetlands, where urbanization pressure is usually most intense. The review will conclude with a prelim-

Table 1. Possible effects of urbanization on wetland hydrology and geomorphology (modified from Ehrenfeld 2000)

Hydrology
- Decreased surface storage of stormwater results in increased surface run-off (resulting in increased surface water input to wetland)
- Increased stormwater discharge relative to base-flow discharge results in increased erosive force within stream channels, which results in increased sediment inputs to recipient coastal systems
- Changes occur in water quality (increased turbidity, increased nutrients, metals, organic pollutants, decreased oxygen concentration etc.)
- Culvert, outfalls etc. replace low-order streams; this results in more variable base-flow and low-flow conditions
- Decreased groundwater recharge results in decreased groundwater flow, which reduces base flow and may eliminate dry-season stream flow
- Increased flood frequency and magnitude result in more scour of wetland surface
- Increase in range of flow rates (low flows are diminished; high flows are augmented) may deprive wetlands of water during dry weather
- Greater regulation of flows decreases magnitude of spring flush

Geomorphology
- Decreased sinuosity of wetland/upland edge reduces amount of ecotone habitat
- Decreased sinuosity and river channels results in increased velocity of stream water discharge to receiving wetlands
- Alterations in shape and slopes (e.g. convexity) affects water-gathering or waste-disseminating properties
- Increased cross-sectional area of stream channels (due to erosional effects of increased flood peak flow) increases erosion along banks

inary functional model linking the major identified drivers and ecosystem condition.

DIRECT HABITAT ALTERATION

Direct habitat destruction and alteration are two of the main causes of global coastal wetland decline. Urban centres have often developed in estuaries and today few of these remain unaffected by human activities (Lindegarth & Hiskin 2001; Heap *et al.* 2001). Upland deforestation due to urban development increases soil erosion within catchments and thus, increases the sediment load entering receiving waterways leading to wetlands (Costanza & Greer 1998). Fragmentation of wetland habitats has been observed to have detrimental impacts on flora and fauna, causing changes in community composition and ecosystem function (Faulkner 2004). These impacts have been associated with changes in habitat patch size and, thus, changes in patch area : edge ratio, resulting in increased edge effects. For example, an increased area of forest edge to forest interior can create greater opportunity for invasive weed species to occupy an area of wetland habitat. Fragmentation of wetland vegetation in the north-eastern USA was correlated with changes in shore bird community structure (Allen & O'Connor 2000).

Fragmentation of wetland habitats can also impact the fauna that depend on these ecosystems for habitat and food, particularly those with specific needs (Schiller & Horn 1997). Observed changes in community structure of shore bird assemblages in the north-eastern USA were consistent with declines in forest interior relative to edge, caused by fragmentation from surrounding residential and urban development (Allen & O'Connor 2000). As a result of urbanization, wetland areas are also affected at the 'complex level', through drainage modification and at the individual level through modification, isolation or fragmentation. Conversely, aquatic organisms may demonstrate different patterns of effect from wetland fragmentation, as many recent studies suggest that it is the interface between vegetated and unvegetated areas, that is, the 'edge' of coastal wetlands that provide the greatest attraction to organisms such as crustaceans and fish (e.g. Vance *et al.* 2002). Predation rates of shellfish in seagrass beds are dependent on the patch size to perimeter ratio (Irlandi *et al.* 1999).

Development of land close to existing wetlands often involves the disturbance of acid sulphate soils (ASS) that either are or once were part of a wetland. After ASS have been oxidized, rainfall results in the flushing of acid from the soil. Run-off helps transport the acid to local waterways. This exposes marine organisms to rapid changes in pH, hypoxia, toxic levels of aluminium and manganese, iron precipitation and hydrogen sulphide. A recent study comparing the release of toxic metals from ASS with industrial effluents has found that such leacheates carry comparable concentrations of Al, Cd, Co, Mn, Ni and Zn to the aquatic environment (Sundström *et al.* 2002). Major environmental impacts caused by ASS within waterways include fish kills and diseases, habitat degradation and changes to aquatic plant communities (Cook *et al.* 2000). Recent estimates suggest that globally approximately 24 million ha of coasts are affected by an acid sulphate problem (Ritsema *et al.* 2000), with large areas of a similar nature already developed for urbanization or other human uses.

Conversion into ponds for finfish and prawn aquaculture, an indirect effect of urbanization, has reduced coastal mangrove area by >50% in most south-east Asian countries (Wilkinson *et al.* 1994). Excavation of natural wetlands for conversion into aquaculture ponds often trigger acid sulphate problems. In developed countries such as Australia, wetland loss around urban centres is mainly through reclamation for residential use, such as canal estates. In south-east Queensland alone, it has been estimated that >1200 ha of mangroves and almost 600 ha of saltmarsh-claypan areas were lost between 1974 and 1987 (Hyland & Butler 1988). During the same period, the area of artificial waterways and canals increased to about 5% of the area of natural mangroves and saltmarsh-claypans in the region. Artificial waterways replacing wetlands create new habitats for selected estuarine fauna, as surveys suggest that the canals support different fish assemblages compared with natural wetlands (Morton 1989, 1992), which may also have different trophodynamics (Connolly 2003).

More subtle effects of the indirect disturbances of urbanization on wetland landscapes may also exist. For example, runnelling that involves constructing shallow wide channels to increase tidal flushing of coastal wetlands is a technique for effective mosquito control in urbanized soft-sediment coasts (Hulsman *et al.* 1989). Through modifying the hydrological regime, runnelling affects wetland community structure such as the burrow size and density of grapsid crabs (Breitfuss *et al.* 2004) as well as reducing mosquito populations. However, long-term research at the Coomera Island reference site in south-east Queensland has shown minimal effect generally on the saltmarsh. Over the first 14 years (since the site was runnnelled in 1985) the vegetation processes were not significantly affected (Dale & Dale 2002). Although mangrove propagules are transported along runnels (Breitfuss *et al.* 2003) the general increase in mangroves in the area is not related to runnelling (Jones *et al.* 2004). With 19.5 years of post-runnelling data the only effects are for slightly increased soil moisture (because tides flood the area more often) and slightly less saline soil moisture, near runnels (because of flushing). There were no effects on the soil or water table pH or on water table depth and salinity (P.E.R. Dale, unpubl. data, 2005). Runnelling is unlikely to lead to acid sulphate problems as it increases wetness and seawater acts as a buffer to acidity (Saffigna & Dale 1999; Alsemgeest *et al.* 2005).

ALTERATION OF THE HYDROLOGICAL REGIME

The hydrological regime is a key determinant in the morphology, species distribution, productivity, sedimentation rates, pollutant transport, nutrient cycling and availability of coastal wetlands (Owen 1995; Hughes *et al.* 1998; Keddy 2000). Conversely, the structure of wetland systems, for example, the type of vegetation cover, is also critical in determining the hydrological balance (Hughes *et al.* 1998). The structure and function of wetland ecosystems are determined primarily by the hydrological regime and its effect on sediment geochemistry (Ellery *et al.* 2003). The tolerance ranges of species to the frequency, depth and duration of inundation exert strong controls on the distribution of flora and fauna within wetlands (Lui *et al.* 2002; Ellery *et al.* 2003). Elements of hydrological management include long-term gradual changes (i.e. climate change and projected sea-level rise) and sudden changes resulting from human interference (i.e. hydraulic modification of tidal flow) (Hughes *et al.* 1998). Direct hydrological changes in wetlands commonly occur as a result of urban development (Pethick 1984; Horner 2000; Lindegarth & Hiskin 2001).

Structural and hydrological modifications (including altered storm water run-off, drainage and filling characteristics) may result in elevated concentrations of trace metals, pesticides, hydrocarbons, sediments and nutrients within the receiving wetland (Nakamura *et al.* 1997; Horner 2000; Lindegarth & Hiskin 2001). Physical changes to landscapes, as a result of massive land movement associated with urban infrastructure construction, also cause geomorphological changes both in the wetlands and adjacent catchment (Nakamura *et al.* 1997).

Groups of water movement (ground flow, surface flow, water flow)

Wetland hydrology involves: (i) groundwater flow; (ii) ground surface flow; (iii) water column flow (tidal water); and (iv) evapotranspiration (Varnell *et al.* 2003). These functional water groups within wetland environments interact with local sediment and nutrients supply to control basin morphology (Varnell *et al.* 2003). Hydrological components are further interactively affected by factors such as surface roughness, topography, dominant vegetation type, pattern of rainfall and tidal range (Nuttle & Harvey 1995; Hughes *et al.* 1998; Darboux *et al.* 2001; Bendjoudi *et al.* 2002). Additionally, wetland hydrology is affected by such factors as land uses within the catchment, wetland to watershed area ratios and system characteristics, for example, sediments, bathymetry, vegetation and inlet and outlet conditions (Horner 2000).

Conversely, the relative importance of these water components in a wetland's hydrological regime would determine the degree of connectedness with adjacent nearshore and terrestrial habitats. Well-connected alluvial corridors within catchment systems characterized by complex surface and subsurface structures provide a wide range of aquatic habitats (Poole *et al.* 2002). Floodplain surface topography also influences the location and composition of vegetative communities on a flood plain. Furthermore, as a control on ground and surface-water exchange, fluvial corridor surface and subsurface morphology can influence biodynamics in fluvial systems by influencing solute transport, carbon and nutrient cycling, and nutrient availability in the channel and hyporheic zone (Poole *et al.* 2002). Flood plain geomorphology is therefore the template on which hydrologic function evolves and the resulting habitat diversity and biocomplexity depend (Poole *et al.* 2002).

Elements of water quality that are affected by changes in wetland hydrology include nutrient transformations, availability, deposition and organic material flux within the system (Whitehouse *et al.* 2000). Net circulation is responsible for exporting organic matter from intertidal mangrove wetlands (Wolanski 1992). Greater surface run-off is also likely to increase the velocity of inflow to wetlands, which can potentially disturb the resident biota and scour substrata. Sediment retention within coastal wetlands and estuaries is directly related to flow characteristics, including the degree and pattern of channellization, flow velocities and storm surges (Horner 2000).

Impact of urbanization on the hydrological regime

Hydrological change is the most visible impact of urbanization and strongly influences water quality, in addition to the hydrodynamic variables within the system. In addition to changes in nutrient and sediment levels, more subtle changes, such as the ratio of particulate to dissolved organic matter and its lability may be affected (Hopkinson & Vallino 1995). Urbanization typically increases run-off peak flows and total flow volumes and damages water quality and aesthetic values (Horner 2000), through conversion of wetland soil into impervious surfaces. Alternatively, dam construction and water extraction due to increasing water demand in urbanized areas also affects coastal wetlands by altering the level and frequency of environmental flow. Hydrologic changes can make an area more vulnerable to pollution as increased water depths or frequencies of flood can distribute pollutants more widely. Changes in hydrological flows generally result in sedimentation instream and decreased depths to the extent that vegetation, especially exotics, invades shallower sections. While floodwaters distribute pollutants more widely, floodwaters result in increased dilution of pollutants. It is the relatively smaller but more frequent stormwater flows, which previously infiltrated soils that now regularly disperse pollutants without the benefit of dilution.

Urbanization usually involves the conversion of natural habitats to land uses with impervious surfaces, for example, pedestrian paths and roads (Faulkner 2004). These surfaces block the infiltration of precipitation and cause changes in hydrology, which degrade downstream ecosystems. Impervious surfaces channel sediments and pollutants directly into drainage networks thereby increasing stormwater run-off into receiving wetlands (Faulkner 2004). In addition, pollutants bound to sediments and organic matter (Connell *et al.* 1999) are distributed through these artificial transport systems and into wetland habitats. This impact will be dealt with in greater detail later.

As changes in surface/subsurface water flow become more common with increased land-use change, more information is needed regarding the factors that influence survival and recovery in sensitive forested wetland areas (Ernst & Brooks 2003). Unfortunately, because of the complex nature of the wetland/watershed relationship, there is still a great deal of uncertainty over the hydrologic budgets and functions of different wetland types (Owen 1995). Most of the existing hydrologic studies of wetlands have been conducted in relatively simple systems, for which the components of hydrologic budgets can be estimated (Owen 1995). These include dyked impoundments and wetlands with a distinct inlet and outlet. Lake-edge or creek-side wetlands are inherently more difficult to study because of the difficulty of accurately measuring sheet flow across the wetland surface into or out of the stream or lake. Similarly, shallow, horizontal groundwater flow through substrata with highly variable hydraulic conductivities, such as peat, makes accurate quantification of this component of the hydrologic budget difficult (Owen 1995).

Wetland ecosystems are particularly susceptible to changes in the timing and quantity of water they receive, as this affects both plant community structure and composition. Community changes occur primarily as a result of variation in flood-tolerances among plants and the effects of flooding on growth rates (Ernst & Brooks 2003). Studies have shown that prolonged flooding causes a compositional shift towards more flood-tolerant tree species through the elimination of less flood-tolerant ones (Ernst & Brooks 2003).

Within the water column changes in hydrologic conditions can either directly modify or alter chemical and physical properties, such as nutrient and toxicant availability pH, salinity, dissolved oxygen concentrations, in addition to the degree of substratum anoxia, sediment geochemistry properties and interstitial infauna and flora (Sriyaraj & Shutes 2001).

NUTRIENT LOADS

Urbanization activities can impact nutrient cycling primarily due to changes in hydrology and nutrient loadings. Such changes can alter plant species composition and nutrient cycling patterns. These impacts can, in turn, alter the species richness and abundance of bird, fish and macroinvertebrate populations (Faulkner 2004). Deforestation in association with urban development may also contribute to increased nutrient loads flowing into receiving wetlands as riparian vegetation provides a sink for nutrients and converts them into less harmful substances (Faulkner 2004). Similarly, alteration of the groundwater flow characteristics by urbanization will probably also affect the nutrient dynamics of coastal wetlands. The role of subsurface flow on nutrient dynamics in coastal wetlands is, however, poorly known.

The impact of nutrient enrichment on coastal wetlands will depend strongly on vegetation type and background nutrient levels in the habitats (Morris & Keough 2003). For example, mangroves and salt marshes are often nutrient-limited communities that would benefit from moderate nutrient enrichment (e.g. Feller *et al.* 2003), while seagrasses generally respond negatively to such enrichment (Hemminga & Duarte 2000).

Eutrophication and organic enrichment

Eutrophication is one of the most significant anthropogenic processes in coastal waters (Rosenberg 1985; Nixon 1995). Increased human activity results in increased nutrient and organic matter discharge into coastal wetlands, whether directly from agricultural run-off or indirectly through discharges such as treated effluents. Given the historical trend of human settlement concentrating around estuarine wetlands, anthropogenic nutrients and organic matter have become an increasingly important component of the material budget of urbanized coasts. This is especially true of coastal wetlands, which typically occur on low-energy, depositing shores. Li and Lee (1998) estimated that approximately 50% of all available organic carbon on which the fishery and waterfowl species depend in Deep Bay, an urbanized embayment in Hong Kong, southern China, was derived from anthropogenic sources.

Urbanization exerts a demand for services such as wastewater treatment (Faulkner 2004). Treated sewage effluents contain toxic metals and are high in nutrients, particularly nitrogen and phosphorus. Bioactive substances such as drug residues and endocrine disrupting substances are also major concerns (Depledge & Billinghurst 1999). Discharge of treated sewage effluent increases the nutrient and pollutant loads in receiving water bodies (Rapport *et al.* 1998; Connell *et al.* 1999). Such nutrient fluxes often cause shifts in phytoplankton community and composition culminating in blooms (e.g. Bowen & Valiela 2001) and result in eutrophication when the algal bloom dies and microbial bacteria exert an increased oxygen demand on the ecosystem during decomposition of the dead algae (Rapport *et al.* 1998; Connell *et al.* 1999). Similar blooms in microalgae have been observed to increase turbidity and contribute to the loss of benthic macrophytes. For example, Costanza and Greer (1998) reported loss of seagrass in Chesapeake Bay estuary preceding phytoplankton blooms. This reduction in benthic plant cover destabilizes the benthos and provides a positive feedback loop for sedimentation and turbidity within the system (Costanza & Greer 1998). The effect of increased nutrient loading is exacerbated by modifications to wetland hydrology associated with urbanization of the surrounding environment through dyking and other forms of impoundment to result in restricted tidal flows (Fong & Zedler 2000).

While the effect of eutrophication and organic enrichment on macrobenthic structure may be predictable (Pearson & Rosenberg 1978), the implications of such changes for the beneficial ecosystem services offered by coastal wetlands can be subtle. Lee (2003) reported that waterfowl numbers overwintering in Deep Bay, a eutrophic mangrove-fringed wetland in south China, was correlated positively with water biochemical oxygen demand (BOD) and total nitrogen load. This finding supports the trophic analysis of the same wetland using both the mass balance and stable isotope approaches (Li & Lee 1998; Lee 2000), in that food chains leading to the waterfowl populations are dependent on anthropogenic rather than natural sources in eutrophic environments. Simple degradable organic matter and moderate levels of nutrient enrichment would have an initial beneficial effect on all but the most nutrient-rich communities, but the positive impacts have to be balanced against the risk of the system 'collapsing' on reaching beyond the 'ecotone' point (Pearson & Rosenberg 1978).

TOXIC POLLUTANTS

Pollutants reach wetlands primarily through catchment and stormwater run-off. Urbanized catchments collect large amounts of pollutants, including eroded sediments from construction sites, trace metals and petroleum wastes from roadways and industrial and commercial areas, and nutrients and bacteria from residential areas (Horner 2000). Just as urbanization produces larger quantities of pollutants, it reduces water infiltration capacity, yielding situations more vulnerable to concentrated transport by surface run-off than pollutants from other land uses.

Within the complex web of wetland sediment-water components, the movement, availability and possible toxicity of contaminants are affected by chemical and physical factors that are, in turn, influenced by hydrological parameters that include residence time, tidal flushing, redox, pH and salinity gradients and temperature (Lau & Chu 1999). Seasonal changes in these factors further compound their importance in governing the levels of bioavailable nutrients within the system (Lau & Chu 1999). Coastal wetlands that are subjected to periodic flooding may demonstrate large temporal variability in such hydrological parameters, as sediment-water nutrient exchange is usually heightened during peak flow periods. Modification of the run-off pattern by urbanization will likely have the same effect in causing larger variability in nutrient exchanges than that of natural wetlands.

The efficacy of wetlands in removing pollutants from the upslope surface and groundwater is highly dependent on hydrology (Corbitt & Bowen 1994). In freshwater wetlands, factors such as the frequency and timing of sampling (e.g. high or low flow period) may affect their apparent effect on nutrient concentrations (Fisher & Acreman 2005). For effective removal of pollutants, sheet rather than highly focused flows must occur and advance at a slow velocity and shallow enough depth to allow interaction with the sediment-water interface (Prior & Johnes 2002). Mechanisms responsible for retention of non-toxic contaminants such as nitrogen (N), phosphorus (P) and suspended sediment include denitrification and assimilation of N, precipitation and sorption of P, trapping of sediments and adhering P and uptake by vegetation. The nutrient removal capacity of a wetland decreases with the flow of water through the riparian zone. In addition, high flows may lead to net nutrient release through the flushing of nutrient rich soil water, desorption processes and sediment erosion (Prior & Johnes 2002). Longer residence times experienced during the growth season, however, promote removal processes, particularly denitrification (Prior & Johnes 2002), effectively increasing nutrient retention. For this reason, subsurface flows in wetland ecosystems are often associated with higher rates of removal than surface flows. Upwards trends in nutrient enrichment of surface waters resulting from diffuse sources have produced concerns about their ecological impacts on wetland systems (Prior & Johnes 2002).

Many toxic inorganic and organic pollutants are of great concern to water quality managers owing to their persistence, toxicity and liability to bioaccumulate (Turner & Tyler 1997). The major temporary or ultimate sink for such pollutants in coastal wetlands and estuaries is the sedimentary reservoir, and the definition of the biogeochemical mechanisms by which they adsorb onto, desorb from and repartition among natural, heterogeneous particle populations is essential in order to assess their environmental fate (Dyer 1995; Turner & Tyler 1997; Whitehouse *et al.* 2000). Within intertidal urban wetlands, the prediction of pollutant distribution is further compounded by intense temporal and spatial gradients of reaction controlling variables such as salinity, dissolved oxygen concentration and particle composition, occurring both within the sediment and in the water column during particle suspension (Turner & Tyler 1997).

Pollutants reach wetlands primarily through stormwater run-off. Urbanized catchments collect large amounts of pollutants, including eroded sediments from construction sites, trace metals and petroleum wastes from roadways and industrial and commercial areas, and nutrients and bacteria from residential areas (Horner 2000). At the same time that urbanization produces larger quantities of pollutants, it reduces water infiltration capacities, making water more vulnerable to transport by surface run-off than pollutants from other land uses. Increased surface run-off combined with disturbed soils can accelerate the entrainment of sediments and the transport and deposition of sediments from the catchment into downstream coastal waters and wetlands (Ehrenfeld 2000; Horner 2000).

An increased use of pesticides, heavy metals, cleaning agents and petroleum products occurs in urbanized environments (Costanza & Greer 1998). These chemicals accumulate in the catchment and are either washed off the land or through storm water drains during periods of heavy rainfall into creeks and rivers that flow into wetlands and estuaries (Costanza & Greer 1998), degrading water quality and habitat (Faulkner 2004). With increases in the proportion of impervious surfaces resulting from urbanization, the impact of more concentrated stormwater run-off in the translocation of chemical pollutants to coastal wetlands is a growing concern. A portion of these pollutants can accumulate in the tissues of wetland organisms and biomagnify through the food web (Connell *et al.* 1999).

Physical, chemical and biological processes interact to manipulate the retention, transformation and release of a large variety of sediments and chemical species within intertidal water bodies and their associated behaviours. Additionally, increased peak flows as a result of increased impervious surfaces through urban development transport more sediment to wetlands, which may alter vegetation community structure, impacting both benthic and other animal species dependent on it (Nakamura *et al.* 1997).

Changes in hydrology can also affect nutrient transformations, availability, deposition and the flux of organic materials within a system (Whitehouse *et al.* 2000). Greater surface run-off is also likely to increase the velocity of inflow to wetlands, which can potentially disturb resident biota and scour substrata. The increased run-off impacts are greater on maintaining the values/services of the remnant terrestrial vegetation, resulting in loss of leaf litter and surface soil organic matter from these areas and further adding to instream loads. Sediment retention within coastal wetlands and estuaries is directly related to flow characteristics, including degree and pattern of channellization, flow velocities and storm surges (Ehrenfeld 2000; Horner 2000).

SEDIMENTATION

Hydrological characteristics within wetlands directly influence the rate and degree of sedimentation of solids imported by run-off into a receiving system. Excessive sedimentation as a result of urbanized catchments may alter wetland topography and soils, and, ultimately result in infilling (Horner 2000). Alternatively, elevated flow velocities can scour a wetland's substratum, changing soil composition, leading to a more channellized flow, resulting in greater velocities, increased erosion and a greater concentration of suspended sediment load within the water column (Whitehouse *et al.* 2000).

Deposition of increased sediment loads within estuaries can reduce flow through wetland systems, providing a positive feedback loop for further increases in sediment deposition within the system (Callaway & Zedler 2004). Such alterations in flow have been associated with changes in water salinities, temperatures and oxygen concentration, and, in extreme cases, may contribute to eutrophic conditions (Fong & Zedler 2000) and subsequent fish and invertebrate kills (Callaway & Zedler 2004).

Vegetation plays an important role in the shaping of depositional forms within intertidal wetlands and on shores experiencing estuarine conditions. Algae and seagrasses partially stabilize the surface of tidal flats and salt-tolerant plants (e.g. mangroves) colonize the estuary margins spread across the intertidal zone (Bird 1984).

Colonization by plants such as reeds and mangroves can occur quickly on newly deposited sediments, if permitted by the prevailing hydrological and wave exposure regimes. Such colonization is likely to interact synergistically with reduced flows and increased sediment deposition, providing an additional feedback loop for the deposition of yet higher sediment loads in urban-impacted wetland systems. Rapid changes in coastal wetland vegetation pattern have been documented to result from wetland landscapes experiencing significant anthropogenic modifications (e.g. Lee 1990). In subtidal areas, increased sedimentation also enhances turbidity and can stress benthic macrophytes such as seagrass (Hemminga & Duarte 2000). Alteration in sedimentation regime might have contributed to the recently documented phenomenon of mangrove intrusion into salt marshes in Australia, reportedly caused by increased nutrient and freshwater delivery (Saintilan & Wilton 2001).

While within-habitat macrofaunal structure may not be correlated strongly with small-scale changes in sediment characteristics (Chapman & Tolhurst 2004), changes in vegetation cover or sediment composition often trigger shifts in macrofaunal assemblages (Lui *et al.* 2002; Thrush *et al.* 2003), which are highly responsive to changes in organic matter content, water depth and other sediment characteristics such as pH and redox regimes. Sedimentation from urbanization per se also results in changes to macrofaunal assemblage structure and abundance, as chemicals carried by the sediment along developed coasts have strong effects on them (Inglis & Kross 2000; Ellis *et al.* 2004).

Sedimentation in ponds and wetlands is important, not only for removing the sediment itself, but also for nutrients and contaminants, which readily attach to fine particles (Walker 2001). Suspended matter has a strong tendency to absorb and adsorb other pollutants. Sediments are major contributors in the removal (or sink) of pollutant in wetlands (Turner & Tyler 1997; Whitehouse *et al.* 2000). There have been a number of studies undertaken of sedimentation in wetlands both in situations where there was an essentially constant flow through and where the main flows were due to run-off from storms. In the latter case, the intermittent nature of stormwater wetlands leads to a situation where sediment is more likely to be distributed around the entire basin (Walker 2001).

In order to understand how sedimentation affects the long-term sustainability of coastal wetlands such as mangroves, it is important to understand the sediment dynamics of such systems. The processes controlling sediment dynamics in vegetated wetlands are, however, not fully understood. It is therefore often difficult to define the sediment budget of an individual degraded wetland and the role the budget plays within the dynamics of the larger coastal sedimentary system (Kitheka *et al.* 2003).

In lowland anastomosing wetland systems, sediment deposition from overbank flow is a critical component of lateral connectivity between river channels and their floodplains that sustains riparian ecology and biodiversity, which may significantly reduce a wetland river system's total suspended sediment load (Wolanski 1992). Understanding the relationship between floodplain sediment assemblages, geomorphic processes and land uses is significant for predicting changes in depositional processes that result from initial anthropogenic disturbances and later attempts to rehabilitate habitats in lowland wetland systems.

Urbanization effects on sedimentation

Urbanization generally increases the amount of sediment carried by terrigenous water supply to coastal wetlands. Sediments within the system can remain either in suspension or on the substratum until it is disturbed from a number of sources, including natural causes, such as aquatic fauna activity or resuspension due to tidal flow. Human activity such as boat movements can also contribute to the mechanisms causing resuspension.

Land-use change resulting in increased soil erosion may increase the supply of terrigenous sediments into mangrove wetlands. This supply is usually within the range that can be tolerated by mangrove wetlands and in moderate volumes is usually essential in substratum accretion which helps plants keep pace with sea level rise (Wolanski 1992). However, during periods of high sediment supply, particularly those associated with extremely high river discharges, enormous volumes of terrigenous sediment are usually discharged into mangrove wetlands. This also occurs as a consequence of natural flood events.

Sediment contamination not only threatens biological life through bioconcentration (bioaccumulation from water) and biomagnification (bioaccumulation from food), but also affects the ecosystem contaminant dynamics (Lau & Chu 2000). Additionally, heavy siltation raises the elevation of wetland so that inundation during flood tide is restricted to zones fringing tidal inlets and main channels (Kitheka *et al.* 2003).

Water quality impacts on coastal wetland sediments can eventually threaten the existence of a wetland. Where sediment inputs exceed rates of sediment export and consolidation, a wetland may gradually become filled. Filling by sediment is a particular concern for wetlands in urbanized areas, as many of them and estuaries have an ability to retain great volumes of sediments (Horner 2000). Enhanced sedimentation results in rapid changes in the form and function of many systems. A key issue is the invasion of terrestrial or semiaquatic vegetation in highly sedimented areas. These alter flows, trap nutrients but cause water quality issues when plant material decomposes.

Within urbanized and degraded catchments, catastrophic siltation events have been recorded following extreme rainfall events (Brooke 2002). During such events, large 'slugs' of sediment from urbanized catchments move downstream into coastal water bodies, rapidly infilling channels with coarser material while finer sediment is deposited across much of the system. The finer-grained sediments that were originally deposited in shallow areas are continually remobilized by wind-generated waves, producing chronic turbidity (Brooke 2002) and changes to tidal and current regimes within intertidal wetlands (Bird 1984).

Fine sediments derived from the catchment and produced within the estuary by the decomposition of biota may also flocculate and settle in the margins of the estuary, forming mud flats where there may have formerly been relatively clean sand (Brooke 2002). Additionally, in association with increased rates of sedimentation, the amount of sediment-bound nutrients, for example, total phosphorous, total nitrogen, total carbon, entering estuaries from their catchments may also increase as a result of urban expansion. As a consequence of increased nutrient inputs, infilling is enhanced, even where the volume of terrestrial sediment influx is low, due to the increased amount of organic material accumulating in the estuary. In combination with high turbidity, these pressures can lead to the loss of healthy benthic habitats.

High suspended material inputs can reduce light penetration, dissolved oxygen and overall wetland productivity (Horner 2000). Pollutants bound to sediments (Turner & Tyler 1997; Whitehouse *et al.* 2000) in run-off can interfere with the biological processes of the aquatic flora and fauna, resulting in impaired growth, mortality and changes in community structure (Horner 2000). Ellis *et al.* (2004) documented significant shifts in mangrove health and macrobenthic community structure in response to sedimentation pattern driven by catchment land-use pattern in two estuaries in New Zealand.

Topography and hydrological changes may effect the stability of the benthic substratum through alteration in water current speeds, increasing turbidity and the potential for increased rate of pollutant transportation within the water body in addition to indirectly and directly affecting resident organisms (Horner 2000). Additionally, an increase in the proportion of impervious (sealed) surfaces and the presence of efficient drainage systems has altered and, often increased the flows and pollutant loads carried by stormwater to local waterways (Bingham 1994; Horner 2000). Problems stem from increased surface run-off due to vegetation being less effective in impeding water flow or retaining nutrients and particulates within the catchment.

In coastal marine environments, the chemical transformations that take place at the sediment–water interface determine the cycling of nutrients and pollutants between the sediment substratum and the overlying water column (Spagnoli & Bergamini 1997; Whitehouse *et al.* 2000). The former can constitute either a source of or a sink for nutrients, so that in shallow areas sediments can be one of the major factors, which control the trophic level of the aquatic system (Spagnoli & Bergamini 1997). Many of these chemical reactions are biologically mediated; their relative importance depends on several factors, such as sediment composition, sedimentation rate, hydrodynamics, bioturbation and irrigation, as well as the physical and chemical characteristics of bottom waters (Spagnoli & Bergamini 1997; Francois *et al.* 2002; Nickell *et al.* 2003).

The sediment characteristics within the system have many implications for the health of the overlying water body. When this sediment is stirred up, trace metals, nutrients and organic contaminants are released into the water column limiting the amount of light essential for plant growth (Dyer 1995). Contaminants such as heavy metals, nutrients, microorganics, pesticides and herbicides have a strong tendency to adsorb to fine-grained sediments, thus relating pollutant transport and storage strongly to sediment dynamics.

An understanding of the sediment sources delivered to, stored within and exported from intertidal waters and wetlands is important for a number of environmental issues including maintenance of navigational channels, light availability for primary productivity, reduction of dissolved oxygen concentrations and the transport and accumulation of particle-bound nutrients and contaminants and their eventual transport to the continental shelf (Eyre *et al.* 1998). Most of our understanding of the relative contributions from the various sediment sources, fluxes and storage in estuaries is derived from studies of large temperate northern hemisphere estuaries (Eyre *et al.* 1998).

Sediments within the system can remain in suspension or on the substratum of the water body until it is disturbed by a number of sources, including natural causes, such as aquatic fauna activity or resuspension due to tidal flow. Human activity such as boat movements can also contribute to the mechanisms causing resuspension. Once initially disturbed, sediment is more likely to be resuspended into the water column because of the unstable bed conditions produced following the settling period. The major biological and physical disturbances that impact on intertidal sediments include: tides, waves, storms, run-off events and sediment movement which results from macrofaunal activities (bioturbation, bioirrigation) and anthropogenic factors such as fishing and dredging (Dyer 1995, 1997; Whitehouse *et al.* 2000; Webb & Eyre 2004). The disturbances occur over a wide range of time scales and influence processes over different space scales (Table 2). Disturbances have a net effect in producing environments that, although broadly structured, are patchy and dynamic.

Resuspension processes in coastal systems affect the cycling of sediments, nutrients, carbon and contaminants. The disturbances are ubiquitous processes influenced by a combination of the strength, speed, direction and duration of both winds and currents, respectively, and on topographical characteristics (Shteinman *et al.* 1997). This is especially true in shallow water bodies.

Table 2. Spatial and temporal scales of disturbance of the intertidal environment (modified from Turner & Tyler 1997)

Disturbances	Spatial scale	Temporal scale
Irrigation	µm–mm	s–h
Biogeochemistry	µm–mm	s–h
Bioturbation	mm–m	s–d
Waves	m	s–h
Fishing	m	s–h
Dredging	m–km	h–d
Run-off events	m–km	h–d
Wind/Storms	m–km	s–d
Tides	m–km	h–d
Sedimentation/erosion	µm–mm	h–dc
Introduction of exotic species	m–km	y–dc
Eutrophication	m–km	y–dc
Global change/sea level rise/geomorphology	µm–m	y–dc

d, days; y, years; dc, decades.

STUDYING URBANIZATION IMPACTS ON COASTAL WETLANDS: SOME METHODOLOGICAL ISSUES

Activities associated with urbanization are likely to impact coastal wetlands by altering their hydrological regimes, water quality, organic matter and nutrient sources, and sedimentation patterns, as well as by deforestation of wetland habitats (Faulkner 2004). Odum (1985) first described how ecosystems might respond to natural and anthropogenic stresses, hypothesizing changes in the energetics, nutrient cycling, community structure and other aspects of ecosystem structure and function. Such ideas have since been further developed to generate the paradigm of 'ecosystem health' (Rapport *et al.* 1998). However, assessment of disturbance effects on coastal wetlands at the ecosystem level has been scarce. Hopkinson and Vallino (1995) reviewed the impact of human activities on run-off patterns, and how these might trigger changes in estuarine community metabolism. Rapport *et al.*'s (1998) idea of the 'ecosystem distress syndrome' (EDS) is based on the theory that ecosystem health as a whole declines in response to anthropogenic activities. Ecosystem indicators, such as changes in primary productivity and nutrient sources, reduced species diversity, horizontal nutrient transport, prevalence of disease and parasitism, extinction of habitat specialists and reduced mutualistic interactions between species are measured to assess the health of impacted ecosystems (Rapport 1998). These predictions have, however, seldom been tested. This is not surprising, as even data on ecosystem dynamics of relatively undisturbed systems are rare. Hopkinson (1992) presents one of a few tests of the predictions on ecosystem development pattern by Odum (1969) on undisturbed systems, comparing the effects of openness and vegetation (forested *vs*. marsh) on the metabolism of four freshwater wetlands. Probably due to the high complexity typical of ecosystem level measurements, the study by Hopkinson (1992) was limited in replication (one site only for each combination of openness and vegetation type) and thus generalizing ability. The lack of such 'baseline' data on wetland ecosystem dynamics makes the detection and prediction of stress effects highly difficult.

Figure 1 provides a theoretical framework against which the impact of urbanization on coastal wetland structure and function may be tested. Various drivers, processes and feedback loops have been identified that eventually lead to indicators of ecosystem stress, such as changes in primary productivity sources and level, and species diversity. In addition to the need to construct functional models for coastal wetlands so as to allow hypothesis testing in relation to urbanization impacts, there is also the need to improve the design of such ecosystem level studies. Most past reports on the impact of urbanization address responses of coastal wetlands in a piecemeal manner, with little control and replication in the comparisons that form the basis for 'impact'. Since most studies on the impact of urbanization are 'natural experiments' largely out of the control of the researcher, temporal replication is usually difficult to achieve. Spatial replication, however, is often possible. Studies that are not spatially replicated will not allow unequivocal detection of impact (Green 1979), especially more so when the anticipated variability in the comparison criteria is high, as is expected of highly dynamic systems such as coastal wetland ecosystems. This high variability also requires that multiple reference locations be used (an asymmetric impact study or 'beyond-BACI' design, Underwood 1994) to ensure that no erroneous conclusions are reached simply because of the choice of a single reference location. In a study on the impact of boardwalk construction on mangrove macrofauna, Kelaher *et al*. (1998) demonstrated large differences in response among study locations to the same disturbance, again pointing to the need for spatial replication. Ecosystem level indicators of urbanization effects (e.g. community metabolism pattern) are expected to be more variable than lower level indicators (e.g. sed-

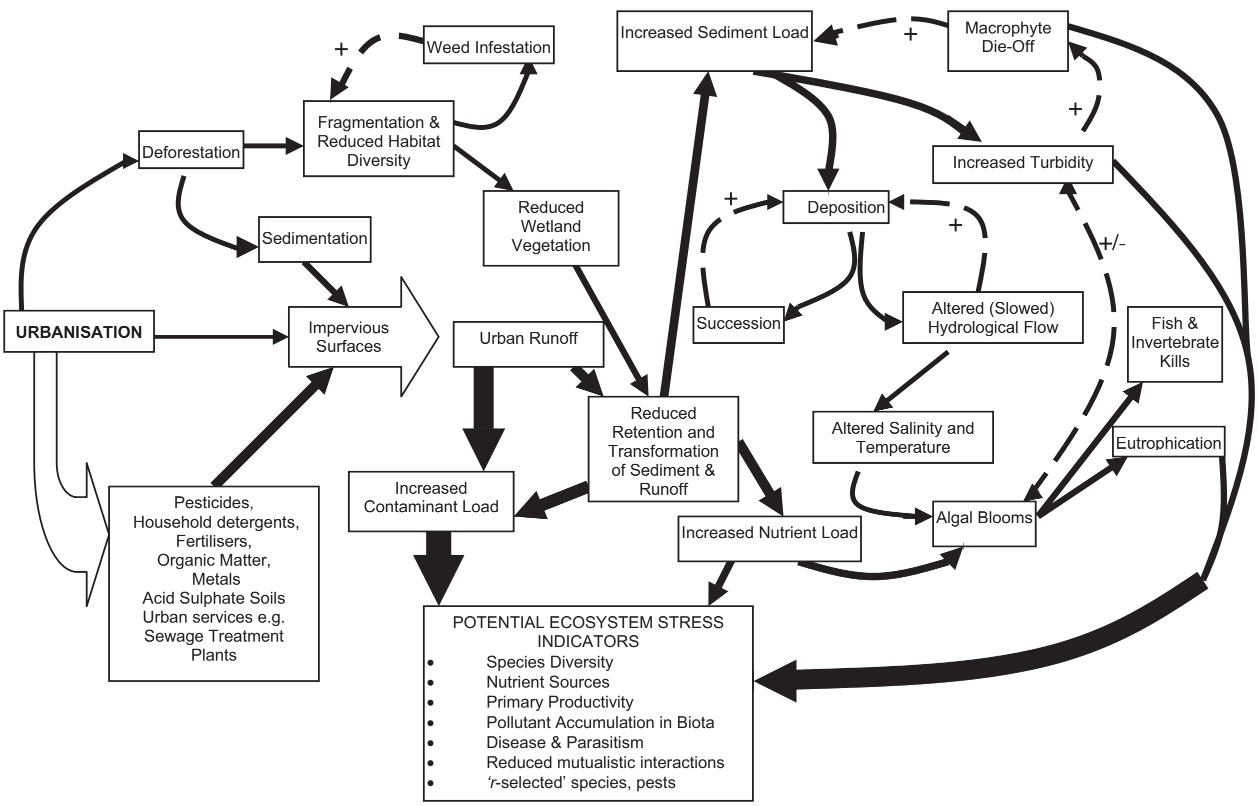

Fig. 1. A summary of the impacts of urbanization on wetland ecosystem function showing disturbances, processes and feedback loops, as outlined in the paper. The strengths of the effects/interactions are indicated approximately by the widths of the lines joining the individual components. Feedback relationships are indicated by dotted lines, with direction indicated by (+/–).

imentation rate) (Connell *et al.* 1999), thus requiring even more extensive replication to confirm impact.

An alternative to the beyond-BACI approach is the use of multiple study locations along a defined urbanization gradient to formulate trends that relate ecosystem level indicators to the degree of urbanization. The application of this approach requires, however, an acceptable metric (e.g. ratio of impervious to pervious surfaces in a catchment) to be devised as an indicator of the level of urbanization. Given the many immediate effects of urbanization on natural ecosystems, the metric would need to incorporate and represent the impacts in a meaningful and unbiased way. This approach has the advantage of not requiring study locations to be categorized rigidly into 'impacted' or 'unimpacted' groups, as today almost no location can really be regarded to be free from human impact.

Furthermore, there is often the practical difficulty in delineating catchments or subcatchments in assessing the extent and impact of urbanization. Traditional elevation models based on broad contour information with typical resolution at approximately 0.5 m are inadequate for identifying hydrologic connectivity through tidal flushing or terrestrial run-off. Digital elevation models constructed using aerial laser surveys offer significantly improved resolution but such information is still far from being widely available. As habitat connectivity through water flow has strong implications for material transport, movement of the biota and general coastal wetland function, the lack of such critical information presents a major barrier to the assessment of the impact of urbanization.

CONCLUSIONS

In addition to causing direct habitat loss, urbanization impacts the structure and function of coastal wetlands through effect on the hydrological and sedimentation regimes and the dynamics of nutrients and pollutants – the major drivers of wetland dynamics. The ecosystem services offered by urbanized wetlands are then compromised by secondary changes in species composition and dominance, habitat connectivity, productivity and metabolism. There is, however, scant unequivocal evidence to support these impacts, mainly because of a lack of knowledge about responses at higher level of organization and methodological issues such as failure to incorporate suitable spatial and temporal replications in impact studies and suitable tools for assessing habitat connectivity.

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