# Urban impacts across realms: making the case for inter-realm monitoring and Management

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## Abstract

Burgeoning populations and the increasing concentration of humans in urban areas result in extensive and increasing degradation and destruction of natural ecosystems. The multitude of impacts and their drivers in urban areas across realms are often studied at local scales, but there is regularly a mismatch between the spatial extent of the impacts and that of the pressures driving those impacts. For example, most human activities occur on land and therefore disturb terrestrial habitats (intrinsic impacts), but their impacts can also extend to the atmosphere and aquatic realms (extrinsic impacts). Management of urban impacts is often designed at local scales and aims to control local pressures, mostly overlooking pressures originating outside the 'managed' area. This is often due to jurisdictional barriers but can also result from the lack of knowledge and recognition of larger scale pressures among scientists and managers. With the aim to highlight the importance of ameliorating extrinsic impacts for holistic management of urban areas, this study discusses the range and extent of extrinsic impacts produced by the most common pressures in urban environments. We discuss that the terrestrial realm is a 'net-donor' of impacts, as most human activities occur on land and the resulting impacts are transferred to aquatic and atmospheric realms. However, activities in aquatic realms can derive in impacts on land. We conclude that, to achieve effective management strategies, greater collaboration is needed between scientists and managers focussing on different realms and regions and we present suggestions for approaches to achieve this.


Keywords: atmosphere; freshwater realm; groundwater realm; marine realm; terrestrial realm; urbanisation

# 1. Introduction

The ecological impacts of urbanisation are extreme and often irreversible, including habitat loss and degradation through pollution, direct modifications (construction) and introductions of non-indigenous species, resulting in ecological change and biotic homogenisation (Walsh et al., 2005; Bulleri, 2006; Lee et al., 2006; Grimm et al., 2008). Even though urban areas at present occupy approximately 1% of the global land area, they are expected to double or triple in size by 2030 (Seto et al., 2011). Most settlements are typically connected to waterbodies by stormwater drainage networks (Walsh et al., 2016b) and many are dependent on groundwater (Morris et al., 2003). Therefore, urbanisation does not only affect the terrestrial and atmospheric realms, but also the freshwater, marine and groundwater realms. When examining the impacts of urbanisation, it is useful to distinguish two main categories of pressure-impact interactions: (i) pressures that arise within a realm, where the impact is also largely constrained to that realm (*Intrinsic*); and (ii) pressures that arise in one realm, but which produce impacts in a different realm (*Extrinsic*).

The recognition of the existence of extrinsic impacts in stormwater management has contributed to a rapid expansion of new integrated approaches that aim to maximize co-benefits for human populations, urban environments and their receiving waters (Fletcher et al., 2015). These approaches include interventions across realms, such as the reduction in impervious areas and protection and enhancement of natural streams and rivers (Fletcher et al., 2015), and provide excellent examples of how to design successful and holistic interventions that consider the connections between realms. Nevertheless, these new approaches have not been applied at scales to achieve their full potential for ecosystem protection (but see Dai et al., 2017), or on other types of extrinsic impacts, such as light and sound pollution and the introduction of pathogens (e.g. Hu et al., 2010; Parris, 2013; Bolton et al., 2017), among others. There are still untapped opportunities for improvement when it comes to recognising the significance and scale of this phenomenon.

In an effort to highlight the relevance and extent of inter-realm impacts, this study proposes to (i) highlight extrinsic impacts derived from pressures originating in the different realms, their direction and type, (ii) consider issues arising from a failure to consider impacts across realms, and (iii) outline avenues to achieve a more holistic understanding of urban impacts across realms in urban areas. For the purpose of this discourse, we define 'realm' as a group of ecosystems that share common physical and ecological attributes and are therefore subject of an area of knowledge or expertise. The 'marine' realm includes all ecosystems present below the high tide mark, including brackish water habitats in estuaries and deltas. The 'freshwater' realm expands from tidal rivers to riparian forests and floodplains, non-riverine wetlands and lakes. Finally, the 'terrestrial' realm includes land-based ecosystems that are only wetted by rain and coastal splash. In making these distinctions we acknowledge that they are conceptual rather than real boundaries, but they provide a simplification to facilitate the general discussion that follows. We also include the realms of atmosphere and groundwater, which may be impacted by extrinsic pressures and also serve as important conduits of pressures between the terrestrial, marine and freshwater realms. While our focus here is on the first three, we have considered the groundwater and atmospheric realms where possible. The extrinsic pressures in the atmosphere and groundwater are extensively documented for some pressures (such as chemical and thermal pollution), but detailed discussion and integration of these realms within this review are limited by the paucity of information on the inherent ecological impacts. Impacts deriving from pressures across realms

This work is the result of discussions held during a workshop attended by a group of 25 experts of each realm in December 2017 in Sydney, Australia. The group identified 11 common pressures (see definition in Panel 1) that connect realms associated with urban areas (Figure 1). These pressures are not acting independently but are highly interconnected and often the result of the same human activities. They all cause a variety of intrinsic (within-realm) and extrinsic (cross-realm) impacts (see definition in Panel 1), but we have focused our discussion here on the extrinsic impacts produced by each pressure, as the impacts within (intrinsic to) a single realm are generally well recognized and have been extensively reviewed elsewhere (e.g. Bulleri, 2006; Lee et al., 2006; Grimm et al., 2008; Attard et al., 2016). It is important to highlight that the strength of extrinsic impacts can vary greatly with the nature of the environment (within realms) and the environmental context of the urban area (across realms). For example, effects of chemical and heat pollution on streams and ponds can be more acute than those experienced by large freshwater lakes which have greater buffering capacity.

### 2.1 Realm conversion

Urbanisation is typified by the modification of the land surface to suit human needs, which can include a transition of habitat from one realm to another. This process may include the conversion of wetlands, streams and marine habitats into dry land and the creation of novel aquatic habitats such as canal estates, reservoirs (impoundments) and artificial wetlands. These changes may be viewed as an extrinsic effect of the habitat being created on the habitat lost. For example, reclaiming mangrove areas increases terrestrial habitats at the expense of marine habitat, with the effect of removing nursery habitat for fish, loss of biodiversity of mangrove/saltmarsh specialists, and loss of the services that transitional habitats such as mangroves provide (Lee et al., 2014). Reclamation of shallow marine habitats for urban or industrial expansion is often associated with the introduction of artificial submerged structures that provide preferred habitats for non-indigenous marine species (Glasby et al., 2007). Conversion of aquatic substrates to terrestrial habitats may also affect groundwater regimes including water level and submarine discharge (Jiao et al., 2001), potentially affecting aquatic communities (see discussion in section 2.3). Similarly, the development of canal estates, which occupy an area of 270 $km^2$ globally (Waltham and Connolly, 2011), creates new aquatic habitat with associated loss of terrestrial habitats, frequently transitional mangrove and saltmarsh habitats.

## 2.2 Hardening

In the terrestrial environment, construction of roads, asphalted/paved urban spaces and buildings replaces permeable with impermeable surfaces. Equivalently, construction of seawalls, breakwaters, piers, and other types of coastal infrastructure, and channelling of streams replaces natural soft and hard substrata with alternative materials that have different chemical and physical properties. In an effort to consolidate terms between realms, we define 'hardening' as the replacement of soil and soft sediment environments by built infrastructure and compaction. This term reconciles the idea of 'impervious surfaces' often used in terrestrial and freshwater literature with that of 'coastal hardening' used in the marine literature.

Hardening produces changes in water movement associated with the changed porosity, morphology and the structural complexity of natural surfaces (Walsh et al., 2005). Low porosity and smoother surfaces of catchments and drainage lines therefore facilitate water runoff into freshwater and marine environments via roads and stormwater drainage networks (Walsh et al., 2016a) and reduce water infiltration and groundwater exchange (Foster, 1988; Sophocleous, 2002). These changes lead to greater volume and intensity of water moving from the terrestrial to the aquatic realm, and the more rapid and voluminous channelling of water from the freshwater to marine realm (Walsh et al., 2012). This run-off can also reduce groundwater recharge rates, or in other cases, result in the addition of contaminants and decreasing quality of groundwater (Foster, 1988).

As the water cycle is a major mechanism of connectivity between realms, any changes in water movements can alter the flow of matter (nutrients, chemical pollutants, etc), organisms and energy between realms, resulting in varied extrinsic impacts. Inter-realm transport of sediments and chemical contaminants is exacerbated by the hardening of flow-paths through the construction of sealed drains (Walsh et al., 2016b). Extrinsic impacts on marine and freshwater realms can result from changes in salinity, chemical contamination and shading (Portnoy et al., 1998; Floerl, 2002; Walsh et al., 2005). The addition of artificial drainage of freshwater habitats such as swamps into estuarine environments has been demonstrated to increase oxygen depletion in estuaries (Johnston et al., 2003). Additionally, hardening in the marine environment, such as construction of seawalls, can have extrinsic impacts on adjacent terrestrial habitats. Seawalls reduce coastal protection (Gedan et al., 2011) and affect the movement of leaf litter between coastal terrestrial and marine environments, changing foraging opportunities for terrestrial organisms (Heerhartz et al., 2014).



## 2.3 Water extraction

The extraction of groundwater for domestic and industrial use in urban areas can lower groundwater tables, leading to extrinsic impacts in the terrestrial environment, such as a loss of phreatophytic (deep rooted) vegetation (Groom et al., 2008). Low groundwater tables and changes in groundwater flows (including reversal) can also lead to a change or loss of connectivity between aquifers and adjoining freshwater water bodies (Sophocleous, 2002). These situations can produce changes in the level (including total drying) and physico-chemical characteristics of surface waters (Sophocleous, 2002), resulting in extrinsic ecological impacts on freshwater systems such as shifts in macrophyte and fish communities (Hayashi and Rosenberry, 2005).

In addition to water from aquifers, many cities draw water from rivers or oceans (via desalination) for human use, albeit usually from areas beyond the urban footprint (Rygaard et al., 2011). The extraction of water from urban aquifers and directly from urban rivers can cause changes in the connectivity between freshwater bodies and groundwater, resulting in extrinsic impacts on the diversity and biomass of groundwater ecosystems (Sophocleous, 2002; Humphreys, 2009) and can potentially reduce flows downstream. The consequences to the marine realm may be changes in the salinity and nutrient dynamics of estuarine and near shore environments that potentially drive extrinsic impacts on marine, coastal diversity and function (Johannes, 1980; Gillanders and Kingsford, 2002). Finally, the use of extracted water in urban irrigation can lead to extrinsic impacts on terrestrial ecosystems, such as shifts to conditions in which native vegetation (particularly in arid zones) have limited ability to colonise and facilitating exotic, hydrophilic species (e.g. Martin et al., 2004).

### 2.4 Chemical contamination

Industrial processes, the application of pesticides and fertilizers, spills and waste generation in urban areas are sources of contaminants in soils, waters and sediments. The main forms of chemical pollution in urban waters are effluents and solids (Kennish, 1996). These include toxic chemicals (heavy metals, pesticides, hydrocarbons, radioactive material), nutrient enrichment, solid waste (plastics and microplastics), as well as materials washed off motor vehicles (lubricants, tyre and brake wear), roads (bitumen, paint) and buildings (corrosives, coating fragments). The extrinsic impacts of land-derived contamination on aquatic environments have been largely studied. Due to the diversity of contaminants, these extrinsic impacts are numerous and diverse, including habitat loss and degradation due to changes in physico-chemical conditions, resulting in changes in biodiversity and functioning of aquatic habitats (Walsh et al., 2005; Johnston and Mayer-Pinto, 2015).

Contamination is one of the few pressures where the extrinsic effects on the atmospheric and groundwater realms are well documented and the management response involves multiple realms. The reduction in air quality and subsequent effect on atmospheric processes due to contaminants derived from activities on land has been well studied (e.g. Rosenfeld, 2000; Da Silva et al., 2018), and there are many examples of contaminants from the terrestrial realm affecting the structure and function of groundwater ecosystems (e.g. Foster, 1988; Stephenson et al., 2013). These contaminants are often transferred or deposited in marine and freshwater realms (e.g. Portnoy et al., 1998; Roy and Bickerton, 2012; Tipping et al., 2014). Finally, contamination of surface waters can also transport contaminants from surface waters to aquifers or lead to permanent clogging from organic compounds affecting the exchange between groundwater and surface waters. These changes can result in extrinsic impacts on groundwater communities (Sophocleous, 2002).

### 2.5 Changes in sediment regimes

The impacts of sediments arising from terrestrial urban areas on the freshwater and marine realms depend to a large extent on their calibre. Fine sediments (<0.5 mm diameter) are readily suspended in flowing water and can be associated with a wide range of attached pollutants (Taylor and Owens, 2009; Houshmand et al., 2014). While they are suspended, sediments can reduce available light and cause physical abrasion, and when they settle in slower-flowing water (freshwater or marine) they can smother habitat and vegetation, reducing aquatic species diversity and altering ecological functioning (Wood and Armitage, 1997a). Coarse sediments (>0.5 mm), in contrast, tend to be less associated with attached pollutants (Houshmand et al., 2014), and remain associated with stream beds, usually providing benthic habitat and playing an important role in maintenance of channel morphology (Hawley and Vietz, 2016). However, sands from disturbed watersheds have the potential to smother riverine habitats (Davis and Finlayson, 2000) and be transported to marine systems.

These extrinsic impacts on freshwater and marine ecosystems are magnified by the major changes to sediment budgets in urban watersheds. Typically, large volumes of sediment mobilised during construction are transported to downstream waters through efficient hydraulic connection of stormwater drains, resulting in elevated yields during the construction phase (Wolman, 1967). Sediment transport can remain high as stream channels scour in response to increased stormwater flows, and the initial model of Wolman (1967) hypothesised greatly reduced sediment yields following the channel reaching geomorphic equilibrium. However, a recent review found that sediment yields often remain high from established urban areas, in part because the post-construction importation of gravels, soils and other particulates provide an effectively unlimited source of of sediments (Russell et al., 2017). Despite increased yields, the beds of urban channels typically hold reduced loads of coarse sediments because increased flow and channel size increase transport capacity (Vietz et al., 2014).

### 2.6 Heat pollution

The increase in hard substratum, reduced tree canopy and industrial activities result in the concentration and retention of atmospheric and surface heat in urban areas known as the ‘urban heat island’ (Kalnay and Cai, 2003). The intrinsic impacts of urban land surfaces on the urban heat island is relatively well understood, although there is surprisingly little research on how the urban heat island affects the ecology of organisms within the terrestrial realm (Chown and Duffy, 2015). This terrestrial heat pollution can have extrinsic impacts in the atmosphere, where air temperature has been related to the composition and abundance of airborne fungal spores (Hasnain, 1993), and in shallow aquifers, where temperatures are also often raised (Zhu et al., 2010), declines in microbial and invertebrate abundance are likely (Griebler et al., 2016). Although the transfer of heat from terrestrial to aquatic realms via stormwater runoff and the discharge of heated water from cooling systems is well understood (e.g. Bienfang and Johnson, 1980; Herb et al., 2008), the extrinsic impacts of thermal pollution have not been well examined as they are often confounded by chemical pollution (Somers et al., 2016). Thermal effluents from cooling systems have been shown to produce extrinsic impacts in the marine realm, affecting phytoplankton mortality and primary productivity (Briand, 1975; Bienfang and Johnson, 1980), benthic community composition (Vilanova et al., 2004; Steinbeck et al., 2005) and the facilitation of invasive species (Wolf et al., 2014). However, the effects can be greatly reduced where there are high mixing capacities at the outflow site (Zieman and Ferguson Wood, 1975).

## 2.7 Light pollution

Ecological light pollution is the change in timing, intensity and wavelength of light through artificial means (Longcore and Rich, 2004). This can be generated directly by lighted structures, street lights, security and vehicles light, fishing boats, oil platforms, or indirectly through sky glow, which is the light scattered back towards the earth's surface by gas and dust in the atmosphere (Longcore and Rich, 2004). Ecological light pollution in the form of artificial light at night is a global problem, with thousands of coastal cities lighting intertidal areas and the potential for skyglow impacts to extend hundreds of kilometres beyond their source (Depledge et al., 2010).

Artificial light sources can cause shifts in reciprocal aquatic-terrestrial fluxes, producing extrinsic impacts in aquatic environments. For example, the community structure of riparian arthropods and aquatic emergent insects can be affected by urban artificial night lighting (Meyer and Sullivan, 2013). The attraction of emergent midges from nutrientenriched aquatic environments to artificial light can cause nuisance swarms in adjacent terrestrial environments (Ali, 1995), and polarised light can create ecological traps in the life cycle of aquatic organisms (Horváth et al., 2009). In marine environments, extrinsic impacts of artificial light on land include shifts in community composition (e.g. Davies et al., 2015; Bolton et al., 2017), reduced success of nesting and disrupted beach orientation in turtles (Witherington, 1992; Rivas et al., 2015), and altered feeding habits of mammals (Yurk and Trites, 2000; Bird et al., 2004) and fish (Batty et al., 1990; Bolton et al., 2017).

## 2.8 Changes in daytime light exposure

While land-clearing can increase daylight exposure in urban areas, conversely the addition of built infrastructure can create artificial shading. These structures can produce 'perpetual' shading, allowing for no or little direct sun exposure (Burdick and Short, 1999). Decreased daylight exposure can produce community-level changes in structure, processes and function (Irving and Connell, 2002; Dafforn et al., 2012). In marine environments, an increase in shading is mostly produced by marine infrastructure such as piers and jetties (e.g. Sanger et al., 2004; Vasilas et al., 2011), therefore the impacts are mainly intrinsic. In contrast, terrestrial urban areas frequently feature reduced vegetation cover, resulting in an increase in daylight exposure of streams. The shading effect of remaining vegetation can be diminished by the widening of channels caused by excessive urban stormwater runoff (Groffman et al., 2003; Hawley and Vietz, 2015). Extrinsic impacts of daylight exposure on freshwater habitats include increased benthic algal growth, which is often magnified in urban streams by increased concentrations of nutrients, particularly phosphorus and nitrogen (Catford et al., 2007). Benthic algal proliferation can smother benthic habitats, clog water intake structures and degrade water quality by reducing dissolved oxygen and pH, resulting in fish kills (Biggs, 2000).

## 2.9 Noise pollution

Even though noise is a common feature of natural environments, anthropogenic noise is often louder and more frequent (Kight and Swaddle, 2011). Anthropogenic sounds, therefore, often mask natural sounds that animals use for communication, detection of danger or spatial orientation, resulting in community-wide changes in diversity and ecosystem services (Barber et al., 2010; Francis et al., 2012). Noise pollution in marine environments is generated by shipping, recreational boating, acoustic harassment devices, industrial activities such as drilling and sonar (all intrinsic). Pressures producing extrinsic impacts in terrestrial environments are considered negligible (Hildebrand, 2009). Contrary to this, extrinsic impacts of land-based noise have been observed in the freshwater realm. Many species of frogs, for example, will change their vocalisations (pitch or frequency) in response to high levels of road noise (Lengagne, 2008; Parris, 2013), or change the timing of the call to minimise temporal overlap with noise occurring at similar frequencies (e.g. Lengagne, 2008; Kaiser and Hammers, 2009). However, some species, such as the Pacific chorus frogs, are unable to compensate for this noise impact (Nelson et al., 2017). Hence, noise pollution can impact breeding success due to increased metabolic costs and reduced ability to communicate (Parris, 2013; Nelson et al., 2017).

## 2.10 Introduction of invasive species

The high rates of traffic between urban centres and the availability of ‘novel’ habitats provided by built infrastructure means that urban areas are often entry points or provide beachheads for invasive species (Hood and Naiman, 2000; Glasby et al., 2007). In addition, the diversity and frequency of disturbances associated with urban areas provide opportunities for establishment and spread of invasive species (Davis et al., 2000; Airoldi and Bulleri, 2011). Invasive species drive changes in composition, processes and functioning of local assemblages (Mack et al., 2000), and the spread of successful invaders and resulting changes in trait distributions can lead to 'biotic homogenisation' (Rahel, 2002; Iannone et al., 2015). Anthropogenic activities in aquatic environments, such as shipping, can have extrinsic impacts on terrestrial environments, as they can be sources of propagules of invasive terrestrial species transported in ballast water or cargo. For example, the bitou bush, *Chrysanthemoides monilifera*, was first believed to have arrived at the east coast of Australia in ballast water (Downey et al., 2007).

There is also potential for freshwater species to produce changes in water quality that extend downstream into marine environments, potentially affecting biodiversity and productivity. For example, invasive zebra and golden mussels, bivalves with great capacity for filter-feeding, efficiently transfer energy from water column to benthos, causing major changes in local and downstream ecosystems where abundant (Karatayev et al., 2007). Extrinsic impacts of terrestrial species feeding on aquatic flora and fauna can also be found. For example, the introduced Norway rat (*Rattus norvegicus*) was observed to feed on a river snail *Viviparus ater* in the margins of the Ticino River at Pavia, Italy (Parisi and Gandolfi, 1974).

### 2.11 Introduction and increased prevalence of wildlife pathogens and their vectors

Urban areas act as entry points for non-indigenous pathogens that travel around the world due to anthropogenic activities, as explained above. Furthermore, human activities in one realm can spread disease-causing organisms into other realms. For example, pathogens enter coastal aquatic areas via sewage disposal, urban waterways and run-off, posing environmental and health risks and potentially having extrinsic impacts by infecting native fauna (e.g. Foster, 1988; Islam and Tanaka, 2004). In addition, zoonotic diseases may be transferred between realms, with mosquitoes as a major vector for transferring diseases between wildlife and humans living within urban areas (Hu et al., 2010; Crocker et al., 2017). Urban areas can also change the prevalence of pathogens by changing wildlife-parasite interactions, with serious consequences for organisms already stressed by urban environments (LaDeau et al., 2015). Pathogens may be more prevalent in urban areas where species densities are higher. This results in greater rates of encounter between hosts and parasites and/or host stress (increased susceptibility to infection, Bradley and Altizer, 2007).

# 2. Moving towards a holistic understanding of urban ecology across realms

Urban areas and the surrounding land and water have experienced the most extreme losses of native biodiversity and functioning in the world (Ruiz et al., 1997; Marzluff, 2001; Grimm et al., 2008). A complete understanding of the drivers of those changes is necessary to preserve the remaining natural environments and restore or rehabilitate lost and degraded habitats. While traditional research and management strategies are realm-specific, effective mitigation of impact requires that extrinsic pressures are considered alongside intrinsic pressures. In much the same way that the ecosystem services literature is now promoting actions that deliver multiple benefits (Connop et al., 2016), a holistic approach to ecosystem research and management may also prove more cost-effective overall.

In the previous section we explored the pressures in urban areas and their extrinsic impacts across realms. It becomes evident that impacts flow from realm to realm in all directions. Nevertheless, terrestrial environments are ‘net-donors’ of impacts to freshwater and marine realms, which tend to be ‘net-receivers’. The degradation and loss of freshwater and marine habitats are largely a consequence of extrinsic pressures from terrestrial environments. This is expected, as the majority of human activities occur on land and there are clear connections to aquatic realms. In some cases, however, activities in aquatic realms can derive in impacts on land. There is often a spatial, and sometimes temporal, mismatch between the realm where the activities and their associated pressures arise, and where the resulting impacts occur. The high connectivity of aquatic systems means that localised pressures in terrestrial environments have the capacity to impact aquatic realms at larger spatial scales. For these reasons, the sources of impacts need to be considered holistically to a far greater extent than is currently undertaken when designing management strategies. The most effective strategies for the management of natural aquatic resources are likely to incorporate not only actions in the 'receiving' realm, but also address the pressures from the 'donor' realm. Such approaches have been included in existing initiatives to control contaminant inputs in aquatic environments, such as new integrated approaches to urban stormwater management (as described in Section 1) and changes in urban design aiming at reducing volume and quality of runoff (Taylor and Owens, 2009), with highly successful outcomes. However, they have not yet been widely considered for management of other drivers mentioned here.

The pressures we have discussed include those that result from human activities taking place in the local area. On top of these, however, urban environments are threatened with regional and global pressures such as climate change and land use. These can interact with local pressures and possibly have synergistic, or antagonistic, effects on the ecological communities in the area. For example, changes in temperature, salinity and acidity can facilitate the establishment of non-indigenous species that, under current conditions, are limited by environmental factors (Floerl et al., 2013). In addition, the type of land use in peri-urban and rural areas surrounding urban environments can greatly influence the composition of assemblages in urban areas. For example, agricultural surroundings increase the diversity of bird species in urban woodlands in Japan (Morimoto et al., 2006), and the proximity to natural areas in regional towns contributes to higher diversities of frogs and birds in suburban landscapes (Smallbone et al., 2011; Luck et al., 2012). A truly holistic approach to the management of urban areas should not only identify intrinsic and extrinsic impacts, but also local, regional and global drivers of change to maximise outcomes.

# 3. Bridging across disciplinary traditions and jurisdictional boundaries

In this paper we have highlighted the great potential for every realm to experience or deliver extrinsic impacts in other realms, but their prevalence and the magnitude of their ecological impacts remain uncertain in many cases. This might be due to two key reasons. Firstly, urban ecology research began as a subdiscipline within each of the broader disciplines of terrestrial, freshwater and marine ecology, which have a history of segregation in research (Cole, 2005; Paine, 2005; Bulleri, 2006). As a consequence, the study of the boundaries and environmental flows between realms has been limited by disciplinary traditions, influencing the types of studies that are undertaken and the methods that are used to perform those studies (Stenseth et al., 2005; Wall et al., 2005; Webb, 2012). Secondly, the disciplinary distinctions are also reflected in the jurisdictional boundaries related to how we conceptualise and manage each realm (Rijke et al., 2013; Pittman and Armitage, 2016). We echo the call of some scientists (e.g. Stergiou and Browman, 2005; Webb, 2012) to generate strategies to overcome these disciplinary and organisational limitations and recognise opportunities for change, such as:

1. Increased exchange of ideas, methods, questions, discussions and other approaches to research between terrestrial, marine and freshwater ecologists (Stenseth et al., 2005)
2. The need for unified terminology (Cole, 2005; Webb, 2012)
3. Funding and research institutions to foster and facilitate connections between realms (Wall et al., 2005)

4. Train future generations of ecologists across realms (Stergiou and Browman, 2005), as well as future generations of urban designers, built environment professionals, engineers and other decision-makers
5. Publication of data or findings done at jurisdictional levels (Paine, 2005)

The successful, all-round management of urban areas will only be achieved when it is regarded as one interconnected environment, instead of a compartmentalised system. This requires new models of collaboration and innovation, as these transformations will only be achieved with the support from multiple levels in science, industry and government, from individual scientists to founding bodies and publishers.

# 4. Conclusions

This manuscript brings to light the multitude of inter-realm connections and highlights the need for integration of realms in science and management. For example, some aspects of changes in hydrology, such as water flow rates during storms, require amelioration measures in both terrestrial as well as stream engineering projects. The solution is likely to be multi-faceted (e.g. planting vegetation, baffles to slow water loss) and needs a holistic approach to success. While the importance of using an inter-realm approach to understand and manage environmental systems is not a new concept, it does remain far from standard practice in the scientific literature and management approaches. Therefore, while what we propose is not new, we do provide an updated reminder of the importance of this approach and a useful advance in how we might conceptualize urban impacts across multiple realms. Effective management of ecological impacts can only be attained in recognition of the full set of pressures driving those impacts. This can be achieved via increased interaction between scientists, regulators and end-users across realms and by designing integrated monitoring strategies in urban environments located at the interface between realms, which explore pressures occurring in all realms.

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# References

Airoldi, L., Bulleri, F., 2011. Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. Plos One. 6, e22985.

Ali, A., 1995. Nuisance, economic impact and possibilities for control, in: Armitage PD, Cranston PS, Pinder LCV, (Eds.), The Chironomidae: biology and ecology of non-biting midges. Springer Netherlands, Dordrecht, pp. 339-364.

Attard, G., Winiarski, T., Rossier, Y., Eisenlohr, L., 2016. Review: impact of underground structures on the flow of urban groundwater. Hydrogeology Journal. 24, 5-19.

Barber, J.R., Crooks, K.R., Fristrup, K.M., 2010. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology and Evolution. 25, 180-189.

Batty, R.S., Blaxter, J.H.S., Richard, J.M., 1990. Light intensity and the feeding behaviour of herring, *Clupea harengus*. Marine Biology. 107, 383-388.

Bienfang, P., Johnson, W., 1980. Response of subtropical phytoplankton to power plant entrainment. Environmental Pollution Series A, Ecological and Biological. 22, 165-178.

Biggs, B.J.F., 2000. Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. Journal of the North American Benthological Society. 19, 17-31.

Bird, B.L., Branch, L.C., Miller, D.L., 2004. Effects of coastal lighting on foraging behavior of beach mice. Conservation Biology. 18, 1435-1439.

Bolton, D., Mayer-Pinto, M., Clark, G.F., Dafforn, K.A., Brassil, W.A., Becker, A., et al., 2017. Coastal urban lighting has ecological consequences for multiple trophic levels under the sea. Science of The Total Environment. 576, 1-9.

Bradley, C.A., Altizer, S., 2007. Urbanization and the ecology of wildlife diseases. Trends in Ecology and Evolution. 22, 95-102.

Briand, F.J.P., 1975. Effects of power-plant cooling systems on marine phytoplankton. Marine Biology. 33, 135-146.
Bulleri, F., 2006. Is it time for urban ecology to include the marine realm? Trends in Ecology and Evolution. 21, 658-659.
Burdick, D.M., Short, F.T., 1999. The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. Environmental Management. 23, 231-240.
Catford, J.A., Walsh, C.J., Beardall, J., 2007. Catchment urbanization increases benthic microalgal biomass in streams under controlled light conditions. Aquatic Sciences. 69, 511-522.
Chown, S.L., Duffy, G.A., 2015. Thermal physiology and urbanization: perspectives on exit, entry and transformation rules. Functional Ecology. 29, 902-912.
Cole, J.J., 2005. Communication between terrestrial and marine ecologists: loud, sometimes abrasive, but healthy and occasionally useful. Marine Ecology Progress Series. 304, 272-274.
Connop, S., Vandergert, P., Eisenberg, B., Collier, M.J., Nash, C., Clough, J., et al., 2016. Renaturing cities using a regionally-focused biodiversity-led multifunctional benefits approach to urban green infrastructure. Environmental Science & Policy. 62, 99-111.
Crocker, W., Maute, K., Webb, C., French, K., 2017. Mosquito assemblages associated with urban water bodies; implications for pest and public health threats. Landscape and Urban Planning. 162, 115-125.
Da Silva, N., Mailler, S., Drobinski, P., 2018. Aerosol indirect effects on summer precipitation in a regional climate model for the Euro-Mediterranean region. Annales Geophysicae. 36, 321-335.
Dafforn, K.A., Glasby, T.M., Johnston, E.L., 2012. Comparing the invasibility of experimental “reefs” with field observations of natural reefs and artificial structures. Plos One. 7, e38124.
Dai, L., van Rijswick, H.F.M.W., Driessen, P.P.J., Keessen, A.M., 2017. Governance of the Sponge City Programme in China with Wuhan as a case study. International Journal of Water Resources Development. 34, 578-596.
Davies, T.W., Coleman, M., Griffith, K.M., Jenkins, S.R., 2015. Night-time lighting alters the composition of marine epifaunal communities. Biology Letters. 11, 20150080.
Davis, J.A., Finlayson, B., 2000. Sand slugs and stream degradation: the case of the Granite Creeks, north-east Victoria. Cooperative Research Centre for Freshwater Ecology Canberra,
Davis, M.A., Grime, J.P., Thompson, K., 2000. Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology. 88, 528-534.
Depledge, M.H., Godard-Codding, C.A.J., Bowen, R.E., 2010. Light pollution in the sea. Marine Pollution Bulletin. 60, 1383-1385.
Downey, P., Holtkamp, R., Ireson, J., Kwong, R., Swirepik, A., 2007. A review of the *Chrysanthemoides monilifera* biological control program in Australia: 1987-2005. Plant Protection Quarterly. 22, 24-32.
Fletcher, T.D., Shuster, W., Hunt, W.F., Ashley, R., Butler, D., Arthur, S., et al., 2015. SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. Urban Water Journal. 12, 525-542.
Floerl, O. Intracoastal spread of fouling organisms by recreational vessels, 2002. PhD thesis. James Cook University, Townsville, pp. 278.
Floerl, O., Rickard, G., Inglis, G., Roulston, H., 2013. Predicted effects of climate change on potential sources of non-indigenous marine species. Diversity and Distributions. 19, 257-267.

Foster, S.S.D. Impacts of urbanisation on groundwater, 1988. Hydrological Processes and Water Management in Urban Areas. Proceedings of the Duisberg Symposium. 198, pp. 187-208.

Francis, C.D., Kleist, N.J., Ortega, C.P., Cruz, A., 2012. Noise pollution alters ecological services: enhanced pollination and disrupted seed dispersal. Proceedings of the Royal Society B: Biological Sciences. 279, 2727-2735.

Gabrielsen, P., Bosch, P. Environmental indicators: typology and use in reporting, 2003. EEA internal working paper. European Environment Agency, Copenhagen, pp. 20.

Gedan, K.B., Kirwan, M.L., Wolanski, E., Barbier, E.B., Silliman, B.R., 2011. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. Climatic Change. 106, 7-29.

Gillanders, B., Kingsford, M., 2002. Impact of changes in flow of freshwater on estuarine and open coastal habitats and the associated organisms. Oceanography and Marine Biology: an Annual Review. 40, 233-309.

Glasby, T.M., Connell, S.D., Holloway, M.G., Hewitt, C.L., 2007. Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? Marine Biology. 151, 887-895.

Griebler, C., Brielmann, H., Haberer, C.M., Kaschuba, S., Kellermann, C., Stumpp, C., et al., 2016. Potential impacts of geothermal energy use and storage of heat on groundwater quality, biodiversity, and ecosystem processes. Environmental Earth Sciences. 75, 1391.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., et al., 2008. Global change and the ecology of cities. Science. 319, 756-760.

Groffman, P.M., Bain, D.J., Band, L.E., Belt, K.T., Brush, G.S., Grove, J.M., et al., 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment. 1, 315-321.

Groom, P.K., Froend, R.H., Mattiske, E.M., 2008. Impact of groundwater abstraction on a *Banksia* woodland, Swan Coastal Plain, Western Australia. Ecological Management & Restoration. 1, 117-124.

Hasnain, S.M., 1993. Influence of meteorological factors on the air spora. Grana. 32, 184-188.

Hawley, R.J., Vietz, G.J., 2015. Addressing the urban stream disturbance regime. Freshwater Science. 35, 278-292.

Hawley, R.J., Vietz, G.J., 2016. Addressing the urban stream disturbance regime. Freshwater Science. 35, 278–292.

Hayashi, M., Rosenberry, D.O., 2005. Effects of ground water exchange on the hydrology and ecology of surface water. Groundwater. 40, 309-316.

Heerhartz, S.M., Dethier, M.N., Toft, J.D., Cordell, J.R., Ogston, A.S., 2014. Effects of shoreline armoring on beach wrack subsidies to the nearshore ecotone in an estuarine fjord. Estuaries and Coasts. 37, 1256-1268.

Herb, W.R., Janke, B., Mohseni, O., Stefan, H.G., 2008. Thermal pollution of streams by runoff from paved surfaces. Hydrological Processes. 22, 987-999.

Hildebrand, J.A., 2009. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series. 395, 5-20.

Hood, W.G., Naiman, R.J., 2000. Vulnerability of riparian zones to invasion by exotic vascular plants. Plant Ecology. 148, 105-114.

Horváth, G., Kriska, G., Malik, P., Robertson, B., 2009. Polarized light pollution: a new kind of ecological photopollution. Frontiers in Ecology and the Environment. 7, 317-325.

Houshmand, A., Vietz, G.J., Hatt, B.E. Improving urban stream condition by redirecting sediments: A review of associated contaminants, 2014. Australian Stream Management conference, Catchments to Coast, June. 30, pp. 549-557.

Hu, W., Mengersen, K., Dale, P., Tong, S., 2010. Difference in mosquito species (Diptera: Culicidae) and the transmission of Ross River virus between coastline and inland areas in Brisbane, Australia. Environmental Entomology. 39, 88-97.

Humphreys, W.F., 2009. Hydrogeology and groundwater ecology: does each inform the other? Hydrogeology Journal. 17, 5-21.

Iannone, B.V., Potter, K.M., Guo, Q., Liebhold, A.M., Pijanowski, B.C., Oswalt, C.M., et al., 2015. Biological invasion hotspots: a trait-based perspective reveals new sub-continental patterns. Ecography. 39, 961-969.

Irving, A.D., Connell, S.D., 2002. Sedimentation and light penetration interact to maintain heterogeneity of subtidal habitats algal versus invertebrate dominated assemblages. Marine Ecology Progress Series. 245, 83-91.

Islam, S.M., Tanaka, M., 2004. Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. Marine Pollution Bulletin. 48, 624-649.

Jiao, J.J., Nandy, S., Li, H., 2001. Analytical studies on the impact of land reclamation on ground water flow. Groundwater. 39, 912-920.

Johannes, R.E., 1980. The ecological significance of the submarine discharge of groundwater. Marine Ecology Progress Series. 3, 365-373.

Johnston, E.L., Mayer-Pinto, M., 2015. Pollution: effects of chemical contaminants and debris, in: Frid CLJ, Crowe TP, (Eds.), Marine Ecosystems: Human Impacts on Biodiversity, Functioning and Services. Cambridge University Press, Cambridge, pp. 244-273.

Johnston, S.G., Slavich, P.G., Sullivan, L.A., Hirst, P., 2003. Artificial drainage of floodwaters from sulfidic backswamps: effects on deoxygenation in an Australian estuary. Marine and Freshwater Research. 54, 781-795.

Kaiser, K., Hammers, J., 2009. The effect of anthropogenic noise on male advertisement call rate in the neotropical treefrog, *Dendropsophus triangulum*. Behaviour. 146, 1053-1069.

Kalnay, E., Cai, M., 2003. Impact of urbanization and land-use change on climate. Nature. 423, 528.

Karatayev, A.Y., Boltovskoy, D., Padilla, D.K., Burlakova, L.E., 2007. The invasive bivalves *Dreissena polymorpha* and *Limnoperna fortunei*: parallels, contrasts, potential spread and invasion impacts. Journal of Shellfish Research. 26, 205-213.

Kennish, M.J., 1996. Practical handbook of estuarine and marine pollution. Vol 10. CRC press, Boca Raton

Kight, C.R., Swaddle, J.P., 2011. How and why environmental noise impacts animals: an integrative, mechanistic review. Ecology Letters. 14, 1052-1061.

LaDeau, S.L., Allan, B.F., Leisnham, P.T., Levy, M.Z., 2015. The ecological foundations of transmission potential and vector-borne disease in urban landscapes. Functional Ecology. 29, 889-901.

Lee, S.Y., Dunn, R.J.K., Young, R.A., Connolly, R.M., Dale, P.E.R., Dehayr, R., et al., 2006. Impact of urbanization on coastal wetland structure and function. Austral Ecology. 31, 149-163.

Lee, S.Y., Primavera, J.H., Dahdouh-Guebas, F., McKee, K., Bosire, J.O., Cannicci, S., et al., 2014. Ecological role and services of tropical mangrove ecosystems: a reassessment. Global Ecology and Biogeography. 23, 726-743.

Lengagne, T., 2008. Traffic noise affects communication behaviour in a breeding anuran, *Hyla arborea*. Biological Conservation. 141, 2023-2031.

Longcore, T., Rich, C., 2004. Ecological light pollution. Frontiers in Ecology and the Environment. 2, 191-198.

Luck, G.W., Smallbone, L.T., Sheffield, K.J., 2012. Environmental and socio-economic factors related to urban bird communities. Austral Ecology. 38, 111-120.

Mack, R.N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M., Bazzaz, F.A., 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecological applications. 10, 689-710.

Martin, C.A., Warren, P.S., Kinzig, A.P., 2004. Neighborhood socioeconomic status is a useful predictor of perennial landscape vegetation in residential neighborhoods and embedded small parks of Phoenix, AZ. Landscape and Urban Planning. 69, 355-368.

Marzluff, J.M., 2001. Worldwide urbanization and its effects on birds, in: Marzluff JM, Bowman R, Donnelly R, (Eds.), Avian Ecology and Conservation in an Urbanizing World. Springer US, Boston, MA, pp. 19-47.

Meyer, L.A., Sullivan, S.M.P., 2013. Bright lights, big city: influences of ecological light pollution on reciprocal stream–riparian invertebrate fluxes. Ecological Applications. 23, 1322-1330.

Morimoto, T., Katoh, K., Yamaura, Y., Watanabe, S., 2006. Can surrounding land cover influence the avifauna in urban/suburban woodlands in Japan? Landscape and Urban Planning. 75, 143-154.

Morris, B.L., Lawrence, A.R., Chilton, P., Adams, B., Calow, R.C., Klinck, B.A., 2003. Groundwater and its susceptibility to degradation: a global assessment of the problem and options for management. Vol 3. United Nations Environment Programme,

Nelson, D.V., Klinck, H., Carbaugh-Rutland, A., Mathis, C.L., Morzillo, A.T., Garcia, T.S., 2017. Calling at the highway: The spatiotemporal constraint of road noise on Pacific chorus frog communication. Ecology and Evolution. 7, 429-440.

Oesterwind, D., Rau, A., Zaiko, A., 2016. Drivers and pressures – Untangling the terms commonly used in marine science and policy. Journal of Environmental Management. 181, 8-15.

Paine, R.T., 2005. Cross environment talk in ecology fact or fantasy? Marine Ecology Progress Series. 304, 280-283.

Parisi, V., Gandolfi, G., 1974. Further aspects of the predation by rats on various mollusc species. Bolletino di Zoologia. 41, 87-106.

Parris, K., 2013. Anthropogenic noise constrains acoustic communication in urban-dwelling frogs. Proceedings of Meetings on Acoustics. 19, 010055.

Pittman, J., Armitage, D., 2016. Governance across the land-sea interface: a systematic review. Environmental Science and Policy. 64, 9-17.

Portnoy, J.W., Nowicki, B.L., Roman, C.T., Urish, D.W., 1998. The discharge of nitrate-contaminated groundwater from developed shoreline to marsh-fringed estuary. Water Resources Research. 34, 3095-3104.

Rahel, F.J., 2002. Homogenization of freshwater faunas. Annual Review of Ecology and Systematics. 33, 291-315.

Rijke, J., Farrelly, M., Brown, R., Zevenbergen, C., 2013. Configuring transformative governance to enhance resilient urban water systems. Environmental Science and Policy. 25, 62-72.

Rivas, M.L., Tomillo, P.S., Uribeondo, J.D., Marco, A., 2015. Leatherback hatchling sea-finding in response to artificial lighting: interaction between wavelength and moonlight. Journal of Experimental Marine Biology and Ecology. 463, 143-149.

Rosenfeld, D., 2000. Suppression of rain and snow by urban and industrial air pollution. Science. 287, 1793-1796.

Roy, J.W., Bickerton, G., 2012. Toxic groundwater contaminants: an overlooked contributor to urban stream syndrome? Environmental Science and Technology. 46, 729-736.

Ruiz, G.M., Carlton, J.T., Grosholz, E.D., Hines, A.H., 1997. Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. American Zoologist. 37, 621-632.

Russell, K.L., Vietz, G.J., Fletcher, T.D., 2017. Global sediment yields from urban and urbanizing watersheds. Earth-Science Reviews.

Rygaard, M., Binning, P.J., Albrechtsen, H.J., 2011. Increasing urban water self-sufficiency: new era, new challenges. J Environ Manage. 92, 185-94.

Sanger, D.M., Holland, A.F., Gainey, C., 2004. Cumulative impacts of dock shading on *Spartina alterniflora* in South Carolina estuaries. Environmental Management. 33, 741-748.

Seto, K.C., Fragkias, M., Güneralp, B., Reilly, M.K., 2011. A meta-analysis of global urban land expansion. Plos One. 6, e23777.

Smallbone, L.T., Luck, G.W., Wassens, S., 2011. Anuran species in urban landscapes: relationships with biophysical, built environment and socio-economic factors. Landscape and Urban Planning. 101, 43-51.

Somers, K.A., Bernhardt, E.S., McGlynn, B.L., Urban, D.L., 2016. Downstream dissipation of storm flow heat pulses: a case study and its landscape-level implications. Journal of the American Water Resources Association. 52, 281-297.

Sophocleous, M., 2002. Interactions between groundwater and surface water: the state of the science. Hydrogeology Journal. 10, 52-67.

Steinbeck, J.R., Schiel, D.R., Foster, M.S., 2005. Detecting long-term change in complex communities: a case study from the rocky intertidal zone. Ecological Applications. 15, 1813-1832.

Stenseth, N.C., Mysterud, A., Durant, J., xeb, l, M., Hjermann, D., et al., 2005. Uniting ecologists into a smooth, tasty and potent blend. Marine Ecology Progress Series. 304, 289-292.

Stephenson, S., Chariton, A.A., Holley, M.P., O'Sullivan, M., Gillings, M.R., Hose, G.C., 2013. Changes in prokaryote and eukaryote assemblages along a gradient of hydrocarbon contamination in groundwater. Geomicrobiology Journal. 30, 623-634.

Stergiou, K.I., Browman, H.I., 2005. Imbalances in the reporting and teaching of ecology from limnetic, oceanic and terrestrial eco-domains. Marine Ecology Progress Series. 304, 292-297.

Taylor, K.G., Owens, P.N., 2009. Sediments in urban river basins: a review of sediment–contaminant dynamics in an environmental system conditioned by human activities. Journal of Soils and Sediments. 9, 281-303.

Tipping, E., Benham, S., Boyle, J.F., Crow, P., Davies, J., Fischer, U., et al., 2014. Atmospheric deposition of phosphorus to land and freshwater. Environmental Science: Processes and Impacts. 16, 1608-1617.

Vasilas, B., Bowman, J., Rogerson, A., Chirnside, A., Ritter, W., 2011. Environmental impact of long piers on tidal marshes in Maryland - Vegetation, soil, and marsh surface effects. Wetlands. 31, 423-431.

Vietz, G.J., Sammonds, M.J., Fletcher, T.D., Walsh, C.J., Rutherfurd, I.D., Stewardson, M.J., 2014. Ecologically relevant geomorphic attributes of streams are impaired by even low levels of watershed effective imperviousness. Geomorphology. 206, 67–78.

Vilanova, E., Mayer-Pinto, M., Curbelo-Fernandez, M.P., Silva, S.H.G.D., 2004. The impact of a nuclear power plant discharge on the sponge community of a tropical bay (SE

Brazil). Bollettino dei Musei e Degli Istituti Biologici dell'Università di Genova. 68, 647-654.

Wall, D.H., Ayres, E., Behan-Pelletier, V., Covich, A.P., Snelgrove, P.V.R., 2005. Soils, freshwater and marine sediments the need for integrative landscape science. Marine Ecology Progress Series. 304, 302-307.

Walsh, C.J., Booth, D.B., Burns, M.J., Fletcher, T.D., Hale, R.L., Hoang, L.N., et al., 2016a. Principles for urban stormwater management to protect stream ecosystems. Freshwater Science. 35, 398-411.

Walsh, C.J., Fletcher, T.D., Burns, M.J., 2012. Urban Stormwater Runoff: A New Class of Environmental Flow Problem. Plos One. 7, e45814.

Walsh, C.J., Fletcher, T.D., Vietz, G.J., 2016b. Variability in stream ecosystem response to urbanization: unraveling the influences of physiography and urban land and water management. Progress in Physical Geography: Earth and Environment. 40, 714-731.

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., Morgan, R.P., 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society. 24, 706-723.

Waltham, N.J., Connolly, R.M., 2011. Global extent and distribution of artificial, residential waterways in estuaries. Estuarine, Coastal and Shelf Science. 94, 192-197.

Webb, T.J., 2012. Marine and terrestrial ecology: unifying concepts, revealing differences. Trends in Ecology and Evolution. 27, 535-541.

Witherington, B.E., 1992. Behavioral responses of nesting sea turtles to artificial lighting. Herpetologica. 48, 31-39.

Wolf, M.A., Sfriso, A., Moro, I., 2014. Thermal pollution and settlement of new tropical alien species: the case of *Grateloupia yinggehaiensis* (Rhodophyta) in the Venice Lagoon. Estuarine, Coastal and Shelf Science. 147, 11-16.

Wolman, M.G., 1967. A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler. 49A, 385–395.

Wood, P.J., Armitage, P.D., 1997a. Biological effects of fine sediment in the lotic environment. Environmental Management. 21, 210–217.

Wood, P.J., Armitage, P.D., 1997b. Biological effects of fine sediment in the lotic environment. Environmental Management. 21, 203-217.

Yurk, H., Trites, A.W., 2000. Experimental attempts to reduce predation by harbor seals on out-migrating juvenile salmonids. Transactions of the American Fisheries Society. 129, 1360-1366.

Zhu, K., Blum, P., Ferguson, G., Balke, K.-D., Bayer, P., 2010. The geothermal potential of urban heat islands. Environmental Research Letters. 5, 044002.

Zieman, J.C., Ferguson Wood, E.J., 1975. Effects of thermal pollution on tropical-type estuaries, with emphasis on Biscayne Bay, Florida, in: Wood EJF, Johannes RE, (Eds.), Elsevier Oceanography Series, Vol 12. Elsevier, pp. 75-98.

# Figures

Figure 1. Direction and type of extrinsic ecological impacts in the various realms (T: terrestrial; F: freshwater; M: marine; GW: groundwater; A: atmosphere) resulting from each of the pressures generated by urbanisation. Biod: loss of biodiversity; CProtect: loss of coastal protection; FWeb: changes in food webs and other food resources; Pop: changes in population attributes; Prod: changes in productivity, NCycle: changes in nutrient cycling. Dashed lines represent potential effects, where there is limited evidence to date.

# Panel 1. Definitions

The terms 'pressure' and 'impact' are commonly used throughout the ecological literature, but not always consistently, even within a single realm (Oesterwind et al.,2016). For this paper, we will be drawing upon the Driver-Pressure-State-Impact-Response (DPSIR) framework outlined by (Gabrielsen and Bosch, 2003; Oesterwind et al., 2016). Urbanisation is a *driver* or process that results in a series of *pressures* (e.g., waste water discharges, artificial surfaces, human behaviours) which act on and contribute to changes in the environments within and surrounding the modified area. *Impacts* are the ecological consequences of the change in environment resulting from a *pressure*. These impacts can be described ecologically at the level of individuals (e.g., changes in feeding behaviour, changes in longevity), populations (e.g., changes in genotypic or phenotypic frequencies, or breeding success rates) and communities (e.g. decreases in biomass, increases in primary productivity, shifts in functional trait distributions). If the magnitude of an impact is sufficient, it can elicit a *response*, such as a change in policy, or investment in actions that either aim to counter the impact (e.g., ex situ conservation actions) or work towards addressing the *pressure* responsible for the *impact* (e.g., investing in upgrading lighting fixtures and light sources to reduce the impact of artificial light at night). We focus here on the *pressures* and *impacts*, and briefly touch upon the opportunity for more coordinated *responses* at the end of our paper.

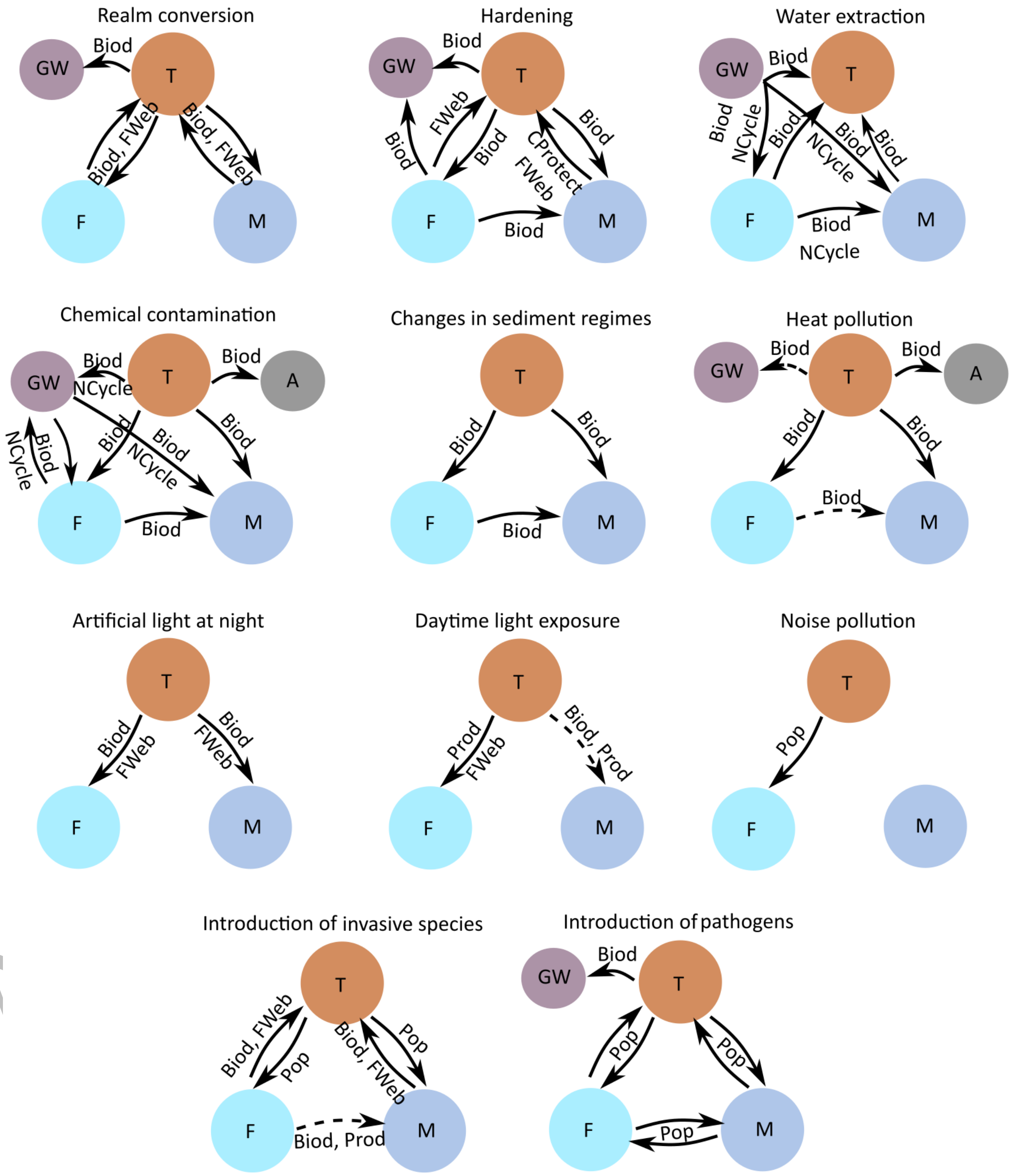


Figure 1