From: "Simon Roberts" <simonroberts2@gmail.com>
Sent: Tue, 31 May 2022 16:25:32 +1000
To: "Information Management" <HVC@huonvalley.tas.gov.au>
Subject: LPS submission Simon Roberts Stormwater SAP
Attachments: Huon LPS submission Simon Roberts Stormwater May 22.docx, Roberts (2021) Review of residential development on the ecological health of receiving waters.docx

Please find attached my submission to the Huon Valley LPS regarding a Stormwater SAP with accompanying document.

Regards,
Simon Roberts & Joanne Wheat

To: The General Manager, Huon Valley Council

Re: Draft Huon Valley Local Provisions Schedule

From: Simon Roberts, 91 Lords Road, Pelverata. 7150

1. Introduction

This report looks at potential nutrient and toxicant issues of aquatic systems arising from residential development in rural areas (often referred to as exurban development) and townships. There is a trend of expanding exurban development in Australia driven by the desire for both amenity and lifestyle changes. Increasing residential development has led to concern about potential degradation of ecological values in rural areas and in particular the impact on waterways and the coastal environment (Tasmanian Planning Commission 2009). Similarly the desire to live in a coastal location has led to increased pressure to expand existing townships within the coastal zone which has the potential to lead to ecological degradation of adjacent water bodies and the marine environment (Victorian Coastal Council et al. 2011).

It has been recognised for some time that changes in land use can have profound and often irreversible impacts on both freshwater and estuarine systems. Harris (2001) reported that land clearing in catchments can lead to far reaching “deleterious changes to soil properties, vegetation and surface and ground water quality and quantity”(Harris 2001). Harris (2001) concluded that at 50% vegetation clearance there is a sharp increase in the export of salinity, suspended solids and nutrients to waterways with a corresponding decline in water quality. He also noted that clearing natural vegetation leads to increased runoff with greater stream power which can cut down into the soil and subsoil of watercourses.

Australian catchments have naturally low levels of export of nutrients to waterways due to low rainfall, generally low relief and low nutrient status of our soils. Freshwater ecosystems, estuarine and coastal lagoons in Australia are therefore particularly susceptible to anthropogenic impacts that can lead to changes in flow or eutrophication (Hadwen and Arthington 2006). Increased nutrient and sediment loads from urban development, waste disposal, agriculture and aquaculture have all been implicated in changes to both river, estuary and coastal lagoon ecology through a deterioration in water quality (Kennish 2002). In general long term water quality monitoring of waterbodies has been restricted to rivers and dams in Tasmania with analysis of land use impacts being mostly attributed to broad scale land use such as grazing, forestry or conservation land (DPIPWE 2020; Hardie and Bobbi 2018; Wagenhoff et al. 2017).

The Resource Management and Planning System (RMPS) of Tasmania has the primary objective of the sustainable development of natural and physical resources and the maintenance of ecological processes. State legislation and State Policies of the RMPS govern the management of freshwater resources and their ecosystems throughout the State. Legislation that contributes to the RMPS shares a common set of high-level objectives (Schedule 1 Objectives of Land Use Planning and Approvals Act 1993). The RMPS also has two State policies that are relevant to protection of both freshwater and marine ecosystems; the Tasmanian State Coastal Policy 1996 and State Policy on Water Quality Management 1997. However, there are few prescriptions within the planning system that consider broadscale ecological impacts of development on aquatic systems.

There is currently a paucity of physical, chemical and benthic invertebrate data from estuaries within the state required to assess the ecological status of these water bodies. This data would be particularly relevant when assessing the potential impacts of current and proposed planning provisions on aquatic environmental values (Edgar, Barrett, and Graddon 1999).

This report details the potential direct and indirect environmental impacts of increased residential development both within and outside established urban zones on waterways (see (Roberts 2021) for a more detailed review on residential land use impacts).

2. Potential direct and indirect environmental impacts of increased residential development on waterways

Increased residential development is a significant driver of decreased aquatic and terrestrial biodiversity (Cuffney et al. 2010; Gagné and Fahrig 2010; King et al. 2011). Urban development or residential development is a considered as one of the most potent land use changes likely to cause degradation to streams on a per area basis (Barmuta et al. 2009; Edgar, Barrett, and Graddon 1999; Urrutiaguer 2016). Increased nutrient, toxicant and sediment loads are highly positively correlated with increases in urban density (Hatt et al. 2004). Edgar *etal* (1999) calculated an “environmental impact factor (EIF)” for natural lands (unmodified vegetated land and water bodies) of 1, an EIF of 5 for cleared forest and an EIF of 20 for urban land. These EIF values are considered to represent the relative increases in nutrient and sediment loads in runoff from each type of land use (Edgar, Barrett, and Graddon 1999). State wide analysis of broad scale effects of land use on 95 environmental factors in Tasmania found that urban land use ranked in the in the top six factors negatively effecting water quality for four of the six indicators examined (DPIPWE 2020).

Current understanding of the impacts of residential development has lead to the realization that a very small area of impervious area as a percentage of total area of a catchment (<2%) can have significant effects on stream ecology (Urrutiaguer 2016). There is also a clear threshold of ~5% catchment imperviousness beyond which ecosystems are substantially damaged (Ewart 2018). In Tasmania urban land use has been implicated in changes in river water quality indicators whilst representing very low levels of the catchment area (DPIPWE 2020). A key message of the DPIPWE (2020) report was the limited information about factors likely to influence river ecosystem health such as the effect of diffuse pollution or temporal changes in land use.

Estuaries and coastal lagoons are considered as particularly susceptible to impacts from changes in land use as they are generally nitrogen limited and are sensitive to increased inputs of nitrogen from fertilizers, urban run-off and land clearing.(Harris 2001) Increased pollution from both point sources (sewage treatment plants, stormwater outfalls) and non-point sources (septic tanks, fertilizer, urban run-off) lead to higher nutrient and organic carbon loading as well as pathogens and chemical contamination of estuarine waters and sediments (Kennish 2002). Urban runoff can have substantially higher concentrations of phosphorus and has a higher pH which can significantly change the vegetation in impacted areas, a common consequence is the establishment of weed species in formally low nutrient soils (Buchanan 1989). Similarly changes in hydrology either as increased or decreased or altered flow regimes can have profound effects on waterways through increased transport of sediments and shifts in pH, salinity and temperature regimes.

Examination of trends in long term datasets of six river health indicators across 85 sites in Tasmania has shown a decline in at least one water quality indicator in 41% of the sites (DPIPWE, 2020). Sites with stable or improving trends were typically at higher elevations (ie higher in the catchment) whereas sites with declining trends were at lower elevations. The impacted sites occurred across all the sampled areas of Tasmania (north, east and south of the state). Differences in trends were attributed to the level of development in catchments with upstream sites generally being undisturbed or with low levels of development. Although few of the sites analysed for long term trends in water quality in Tasmania were in the Huon Valley municipality the general trend of increased development in the lower reaches of catchments is typical of most catchments in the municipality.

Potentially important cumulative or broad scale diffuse effects of development is considered a key consideration for landscape planning in coastal areas (Victorian Coastal Council et al. 2011). In Tasmania other than through local planning schemes there is little integration between the management of catchments and the coastal and marine zones. The recently adopted Rural Water Use Strategy had little consideration of catchment water use on the ecological function of estuarine or coastal ecosystems. The strategy stated that;

> "*Whilst water quality is a consideration in executing functions under the WMA, catchment management and management of water quality more generally are principally managed through other suitable frameworks and instruments outside the water management framework as it relates to the Rural Water Use Strategy."*

The "other suitable frameworks and instruments" are not listed in the Rural Water Use Strategy. Land use planning would be one such mechanism that could be used to control broad scale effects on water quality by limiting potentially threatening types of use or development and designating mitigation actions when uses are potentially threatening to ecological function of waterbodies.

3. Recommendations for avoiding or mitigating impacts from urbanization on waterways and estuaries

Currently there is a lack of recent data on physio-chemical (salinity, flow, temperature, pH), biodiversity, nutrients or toxicants in either the water column or sediments of waterways and estuaries in the Huon. Most of the data collected is more than 10 years old has been opportunistic, limited in extent and has not captured seasonal or annual trends. A near natural flow regime is required to maintain the natural values present in the system (endemic or threatened species, floodplains and riparian communities), however in most of these systems these values have not been assessed with a level of rigour that provides certainty that all the values have been identified.

There is now a general recognition that residential development will lead to increased stormwater run-off with high levels of associated pollutants. Other jurisdictions have implemented mechanisms to try and mitigate or minimise the effect of residential development (and its associated infrastructure) on water bodies. In Victoria there is now state wide guidance from the EPA in relation to urban stormwater (EPA (Vic) 2021). In Victoria residential developments are encouraged to mitigate the amount of stormwater generated through on-site infiltration or use of stormwater as their "general environmental duty". There is also a required reduction in pollutant loads of 45% for nutrients (nitrogen and phosphorus) and 80% for suspended sediment compared to the untreated runoff (EPA (Vic) 2021). The *Tasmania the State Policy on Water Quality Management 1997* requires that:

> *31.1 Planning schemes should require that development proposals with the potential to give rise to off-site polluted stormwater runoff which could cause environmental nuisance or material or serious environmental harm should include, or be required to develop as a condition of approval, stormwater management strategies including appropriate safeguards to reduce the transport of pollutants off-site."; and*
>
> *33.1 Regulatory authorities must require that erosion and stormwater controls are specifically addressed at the design phase of proposals for new developments, and ensure that best practice environmental management is implemented at development sites in accordance with clause 31 of this Policy.*

The current and potential increase in residential development adjacent too and in the catchment of sensitive waterbodies is highly relevant to the implementation of the planning scheme. Protecting the natural flow regime of adjacent and upstream waterways and ensuring good water quality are critical to maintaining their biodiversity and ecological processes. Residential development should as much as possible be restricted to the current serviced townships with appropriate mitigation of stormwater impacts through water sensitive urban design principles (Fletcher et al. 2015).

Water sensitive urban design (WSUD) principles can be implemented in any development that has the potential to change the water balance of a parcel of land through the construction of impervious surfaces and/or artificial drainage. The original aims of WSUD where to (cited in (Fletcher et al. 2015)):

1. manage the water balance (considering groundwater and streamflows, along with flood damage and waterway erosion),
2. maintain and where possible enhance water quality (including sediment, protection of riparian vegetation, and minimise the export of pollutants to surface and groundwaters),
3. encourage water conservation (minimizing the import of potable water supply, through the harvesting of stormwater and the recycling of wastewater, and reductions in irrigation requirements), and
4. maintain water-related environmental and recreational opportunities.

A simpler aim for new developments would be to achieve:

- Natural frequency of surface run-off.
- Natural volumes of run-off.
- Natural infiltration rates.
- Natural concentrations of pollutants

These aims are consistent with objectives of the State Policy on Water Quality Management 1997 and would better protect adjacent and downstream water bodies if implemented for new developments.

Varying levels of stormwater infrastructure are in place in many of the townships of the Huon Valley municipality. Traditionally storm water management has been to convey additional flows generated by increased impervious surfaces to the nearest water course in order to reduce the risk of flooding. In most cases this infrastructure increases the risk of environmental damage by reducing the possibility of infiltration or trapping of sediments if this water had followed a natural flow path over pervious areas. Increased connection to current or planned flood mitigation stormwater infrastructure is therefore likely to be an ongoing threat to adjacent water bodies. Potentially mitigation of some of these impacts from “end of pipe” flows from serviced stormwater areas could be directed to appropriately designed retention systems.

A further consideration is the provision of sewage infrastructure including its proximity to water bodies, level of treatment and risk of overflow or leakage. In areas not serviced by sewage pipes septic tanks are the primary waste water treatment. Risks from septic tank to adjacent water bodies are dependent on the proximity to the water course, type and size of system and level of maintenance. An audit of septic systems to check that they are working properly or require upgrading in areas close to sensitive aquatic assets may be appropriate.

4. Planning as a tool to minimise degradation of aquatic resources

The implementation of the planning scheme should further the objective of protection and or enhancement of the ecological function of waterways consistent with the objectives of Schedule 1 of LUPPA; objectives 1 (c) & (e) of the Water Management Act 1999; objectives 3 (a), (c) & (h) of the Environmental Management and Pollution Control Act 1994; and objectives 6.1 (a), (b) & (d) of the State Policy on Water Quality Management 1997.

Residential development will in many cases be located in the coastal zone. All developments within one kilometer of the coast will be subject to the objectives and principles of the State Coastal Policy 1996 and its outcomes. Of particular relevance are the outcomes;

> *1.1.1 The coastal zone will be managed to ensure sustainability of major ecosystems and natural processes.*
>
> *1.1.5 Water quality in the coastal zone will be improved, protected and enhanced to maintain coastal and marine ecosystems, and to support other values and uses, such as contact recreation, fishing and aquaculture in designated areas.*
>
> *1.1.9. Important coastal wetlands will be identified, protected, repaired and managed so that their full potential for nature conservation and public benefit is realised. Some wetlands will be managed for multiple use, such as recreation and aquaculture, provided conservation values are not compromised.*
>
> *2.1.1. The coastal zone shall be used and developed in a sustainable manner subject to the objectives, principles and outcomes of this Policy. It is acknowledged that there are conservation reserves and other areas within the coastal zone which will not be available for development.*
>
> *2.1.2. Development proposals will be subject to environmental impact assessment as and where required by State legislation including the Environmental Management and Pollution Control Act 1994.*
>
> *2.1.5. The precautionary principle will be applied to development which may pose serious or irreversible environmental damage to ensure that environmental degradation can be avoided, remedied or mitigated. Development proposals shall include strategies to avoid or mitigate potential adverse environmental effects.*
>
> *2.4.1. Care will be taken to minimise, or where possible totally avoid, any impact on environmentally sensitive areas from the expansion of urban and residential areas, including the provision of infrastructure for urban and residential areas.*
>
> *2.4.2. Urban and residential development in the coastal zone will be based on existing towns and townships. Compact and contained planned urban and residential development will be encouraged in order to avoid ribbon development and unrelated cluster developments along the coast.*

2.4.3. Any urban and residential development in the coastal zone, future and existing, will be identified through designation of areas in planning schemes consistent with the objectives, principles and outcomes of this Policy.

There are limited opportunities within the planning scheme to influence changes in land use that may affect water quality. One area where the planning scheme has a significant influence is on the type, size and intensity of residential development and where this may occur. Strategies to manage urban development in undisturbed catchments, such as zoning and land use planning can be important tools to prevent or minimise the degradation of aquatic environments. Similarly planning tools have also been used to initiate stream-rehabilitation efforts that can have a positive effect on the biological condition and health of streams (Coles 2012; Prosser, Morison, and Coleman 2015; Vietz et al. 2016). Using impervious cover (or connected impervious cover) as a surrogate for the many correlated stressors driven by urbanisation has the potential to be used as a planning tool to trigger the implementation of “end of pipe” measures to protect the ecological function of water bodies. Alternately “source control” at the lot or individual development stage using WSUD or other treatment methods to mimic predevelopment conditions is likely to be more effective and consistent with the “user pays” principle. Retrofitting of WSUD measures may also be appropriate when intensification of development is proposed in a semi-developed area.

The most effective method to prevent additional impacts from residential development in sensitive areas is to rezone privately zoned land to zonings where residential use is discretionary and subject to performance standards that will protect or enhance ecological values, the implementation of the new Landscape Conservation zone may give additional control over these threats in the peri-urban areas of the municipality. Similarly zoning that restricts sub-division or encourages consolidation of lots will generally reduce the pressure for additional residential development and its associated additional infrastructure such as roads and services.

A key requirement of both the *State Policy on Water Quality Management 1997* and the *State Stormwater Strategy 2010* are the promotion of source control strategies that treat, store and infiltrate stormwater on-site with an aim of reducing flows and decreasing pollutant concentrations. The *State Policy on Water Quality Management 1997* Clause 33.2 requires that:

> *“State and Local Governments should develop and maintain strategies to encourage the community to reduce stormwater pollution at source.”*

The Huon Valley LPS should include a Stormwater Specific Area Plan that at a minimum applies to all development within the coastal zone and within 40m of a waterway (class 1 to 4 streams, and lakes). It should have an objective that requires; “*That development provides for adequate stormwater management*.”. An acceptable solution would be to (A1) *meet the stormwater quality and quantity management targets identified in the State Stormwater Strategy 2010.* Additionally a performance standard could be (P1) *must treat, store and infiltrate stormwater on-site*.

The generation of additional stormwater from new developments being connected to the existing stormwater infrastructure is likely to be detrimental to many of the aquatic assets of the municipality. Additionally extra flows from developments not connected to the stormwater system are also likely to increase pressures on aquatic habitats.

A key objective of a Stormwater SAP should be to reduce the overall quantity and improve the quality of urban stormwater flows to waterbodies as part of a comprehensive stormwater management program that is premised on the identification of important aquatic ecosystem values and the need to avoid or minimise any potential ecological impacts. A priority should be the management of stormwater to reduce overland flow and to increase water quality at source and where this is impractical then as part of a local treatment process incorporated into the council stormwater infrastructure.

Many studies into the effect of urbanisation on aquatic systems have shown that ecological impacts can occur at very low levels of residential development. Overall impacts of new developments on aquatic systems can be much more effectively managed and lead to less cost if these developments are primarily in already serviced areas and are discouraged in unserviced settlements or in cluster developments outside serviced areas.

References.

All the references in this document can be found in the accompanying report:

Roberts, Simon. 2021. *Review of Residential Development on the Ecological Health of Receiving Waters*.

Review of residential development on the ecological health of receiving waters

Simon Roberts Nov 2021

1. Introduction

This report reviews the current understanding of the impact of residential development on the ecological health of receiving waters. Most of the literature on the effect of urbanisation has focused on impacts at the stream level as this is the most common surface water directly impacted by changes in land use. Many factors contribute to the quality of a stream and how it is affected by residential development. Fundamentally, stream ecological function is controlled by five variables: climate, geology, soils, land use, and vegetation. These variables directly affect two of the key drivers of change in stream function of discharge and sediment load, which in turn has an impact on the hydrology, morphology and ecology of the stream (Brabec et al., 2002). Of these variables, land use and vegetation are generally the only ones that can be controlled through land use planning and are therefore often the focus of studies examining degradation, protection or rehabilitation of streams.

Studies in the late twentieth century tried to define thresholds of urban development (defined by different measures of urbanisation; see below) where ecological impacts occur. Many of these studies concluded that degradation occurred in a continuous rather than at a defined threshold, although there can be distinct break points and for many indicators a maximum level of impact at low or intermediate levels of land use change. Additionally, the concept of degradation at a particular site in a catchment fails to incorporate potential cumulative or synergistic impacts within a catchment that may be missed by studying a single site at the end of a sub-catchment.

More recent studies have examining the ecological impact of increasing urbanisation on the aquatic values of waterways by examining physical and biological changes in catchments across urban to rural gradients. A common feature of these studies is that biological effects are often observed in streams at very low levels of urban development within catchments. Determining the exact mechanisms of degradation is often confounded by the many correlated landscape changes that disrupt the natural biological and geomorphic processes in streams in urbanising catchments. Key drivers of change have been identified as decreased vegetation cover, a reduction in organic material supply, increased impervious areas, more efficient delivery of stormwater to waterways, increased overland flows, increased catchment erosion and increased nutrients and toxicants (Grimm et al., 2008; Sheldon et al., 2012). Additionally it is also recognised that restoration of these values in previously impacted catchments is often complex and expensive (Hughes et al., 2014; Prosser et al., 2015; Urrutiaguer et al., n.d.) even at low levels of development (Walsh et al., 2015).

Urbanisation exerts a disproportionately large influence compared to most other land use changes on steam function (Paul & Meyer, 2001). Degradation of stream ecological function is driven by increased frequency and magnitude of storm flows, increased total flow, reduced dry-weather flows, changes to riparian and in-stream habitat and increased loads of nutrients and toxicants (Paul & Meyer, 2001; Roy et al., 2009; Urrutiaguer, 2016; Walsh, Roy, et al., 2005). All of the principal mechanisms by which land use influences stream ecosystems identified by Allan, (2004) in Table 1 are associated with changes driven by urbanisation.

TABLE 1. Principal mechanisms by which land-use activities influence stream ecosystems. (From Allan 2004.)

Environmental factor	***Effect***
Sedimentation	Increases turbidity, scouring, and abrasion; impairs substrate suitability for periphyton and biofilm production; decreases primary production and food quality causing bottom-up effects through food webs; in-filling of interstitial habitat harms crevice-occupying invertebrates and gravel-spawning fishes; coats gills and respiratory surfaces; reduces stream depth heterogeneity leading to decrease in pool species
Nutrient enrichment	Increases autotrophic biomass and production, resulting in changes to assemblage composition, including proliferation of filamentous algae, particularly if light also increases; accelerates litter breakdown rates and may cause decrease in dissolved oxygen and shift from sensitive species to more tolerant, often nonnative species
Contaminant pollution	Increases heavy metals, synthetics, and toxic organics in suspension, associated with sediments, and in tissues; increases deformities; increases mortality rates and impacts to abundance, drift, and emergence in invertebrates; depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance
Hydrologic alteration	Alters runoff–evapotranspiration balance, causing increases in flood magnitude and frequency, and often lowers base flow; contributes to altered channel dynamics, including increased erosion from channel and surroundings and less-frequent overbank flooding; runoff more efficiently transports nutrients, sediments, and contaminants, thus further degrading instream habitat. Strong effects from impervious surfaces and stormwater conveyance in urban catchments and from drainage systems and soil compaction in agricultural catchments
Riparian clearing/ canopy opening	Reduces shading, causing increases in stream temperatures, light penetration, and plant growth; decreases bank stability, inputs of litter and wood, and removal of nutrients and contaminants; reduces sediment trapping and increases bank and channel erosion; alters quantity and character of dissolved organic carbon reaching streams; lowers retention of benthic organic matter owing to loss of direct input and retention structures; alters trophic structure
Loss of large Woody debris	Reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability; influences invertebrate and fish diversity and community function

2. Measures of urbanisation

In order to study effects on of urbanisation on waterways a measurement of urbanisation intensity is required. It seems logical that a good measure of urbanisation would be residential density, however there is a general pattern of higher amounts of impervious area per residence as urban density decreases (National Research Council, 2009). Where aquatic ecological impact is concerned the percentage impervious cover in a catchment is commonly used as impervious surfaces (local and regional roads, shops, sheds, driveways and utilities) are the main source of increased runoff, which is implicated in many of the direct biotic and abiotic effects on stream function (Arnold & Gibbons, 1996). The proportion of T**otal Impervious** (T**I**) area in a catchment is frequently highly correlated with ecological impacts (Taylor et al., 2004). However some studies have shown that areas of impervious surface directly connected (via pipes or channels), referred to as **Effective Impervious** (**EI**) provides a better fit to some parameters (Hatt et al., 2004). A more sophisticated measure, **Attenuated Impervious** (**AI**) combines both the directly connected surfaces and weights none connected surfaces or ends of pipes according to their distance from the stream. A proxy for directly connected impervious (EI) that is sometimes used is road density, expressed as kilometres of road per square kilometre of land (km/km$_2$) and is considered appropriate as roads are often the main component of EI (Hopkins et al., 2015; National Research Council, 2009).

3. Hydrology

Urbanisation alters the hydrological function of streams in a number of ways (Hopkins et al., 2015; Vietz et al., 2014). The most common affect is larger and more frequent runoff generated flows primarily from the replacement of previously pervious landscapes (forest and grasslands) with impervious urban surfaces that are in close proximity (<50m) or directly connected to streams. These increased runoff events from urban infrastructure (buildings, driveways, local roads) lead to more frequent and higher peak flows that can modify the stream channel either through the delivery of increased sediment loads or through scouring and transport downstream. Increased flows even after small rainfall events can have profound effects on the water balance of catchments by reducing the amount of water that would have infiltrated into the local groundwater leading to reduced base flows during dry periods. Residential development in forested catchments also leads to a reduction in forest area, through clearing for housing and sheds, bushfire mitigation and increased road access. Replacement of forest cover with grassland or urban infrastructure reduces the rate of transpiration and increases the likelihood of surface flows through reduced interception by vegetation. Removal of streamside vegetation can also lead to bank instability and increased incision of the channel that lowers the groundwater level of the riparian zone.

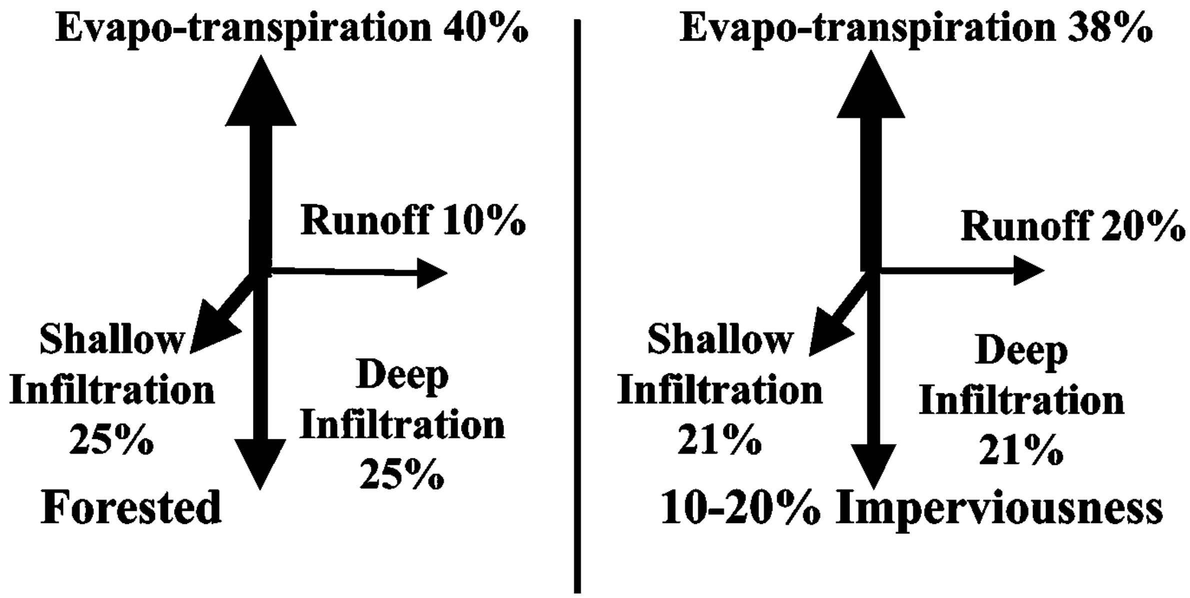

Figure 1. Changes in hydrologic flows with increasing impervious surface cover in urbanizing catchments (after Arnold & Gibbons 1996).

A number of studies have shown linear increases in both the magnitude and frequency of high flow events as the proportion of impervious cover increases in a catchment. Hopkins et al (2015) reported linear increases in high flow events with shorter duration across 8 of 9 urban gradients ranging from 0% to 60% impervious cover in the USA. In Australian cities the volume of runoff is typically 5-10 times the pre-urban volumes (Walsh et al., 2010). Arnold & Gibbens (1996) estimated a doubling in total stream flow with an increase in impervious surfaces from 0% to 20%.(Figure 1). Vietz et al. (2014) studied the effect of increased flow events on geomorphology of streams and estimated that an increase from 0% to 2% EI would increase the duration of discharges likely to transport sediments by 12% in a Melbourne stream. Similarly Vietz et al. (2014) found that urbanisation significantly impacts a number of geomorphic attributes of streams (presence of bars/benches, bank instability and presence of large wood) at EI values <2% which is equivalent to TI of 4-5%. They concluded that measurable geomorphic change occurs at very low levels of EI (0-3%) and that stream management of degradation should focus on stormwater drainage (Vietz et al., 2014). One study found that a small increase in EI to >3% led to streams being almost entirely scoured to bedrock or clay (Sammonds et al. (2014) cited in (Vietz et al., 2016)).

4. Nutrient cycling

Urbanisation rapidly leads to increased loads of nutrients (primarily nitrogen and phosphorus) that are often drivers of eutrophication in fresh and saline waters (Hatt et al., 2004; Lintern et al., 2018; Taylor et al., 2004). Increased nitrogen loads are derived from increased depositional sources associated with urban land use (fertilizers and atmospheric deposition, domestic animal manure (Bettez & Groffman, 2013; Lintern et al., 2018)) which can be efficiently delivered to streams by storm flows through pipes and channels. Septic tanks deliver most of their nitrogen output as soluble nitrate (NO_3) primarily to groundwater which can be delivered to streams through sub-surface flows (Hatt et al., 2004; Walsh & Kunapo, 2009).

Reduced forest and shrub cover leads to decreased assimilation by vegetation and lower levels of supply of wood and organic carbon to streams (Lammers & Bledsoe, 2017). Reduced in stream carbon cycling can decrease nitrogen (and soluble phosphorus) retention times in the terrestrial and aquatic environment (Grimm et al., 2005). Urban derived hydrological and geomorphic changes (less ground water supply and channel incision) can also disrupt groundwater and flowing water interactions in both the riparian and hyporheic zones of the stream which can decrease the natural loss of nitrogen as N_2 gas through denitrification (Lammers & Bledsoe, 2017; McClain et al., 2003).

Increased soluble phosphorus concentrations in streams come from diffuse and point sources associated with urban land use (septics, sewage treatment plants, fertilizers and organic contaminants such as animal wastes). Reduced riparian vegetation decreases in-stream organic carbon which can decrease phosphorus assimilation (Lammers & Bledsoe, 2017). In many Australian soils phosphorus is a limiting nutrient for plant growth, increased phosphorus supply from urban sources generally promotes weeds which are more adapted to higher nutrient soils (Buchanan, 1989). A large amount of terrestrial and aquatic phosphorus is bound to soil and sediments particles, mostly fine sand, clays and silts (Houshmand et al., 2014) and is typically mobilised to streams from increased erosion of pre-existing upland sources (Lovett et al., 2007). The increased power of storm flows in the stream channel also leads to mobilisation of bank and bed sediment which can have high concentrations of particulate phosphorus (Lammers & Bledsoe, 2017). Most of this particulate phosphorus is delivered to aggrading sections of the stream system or downstream receiving waters (lake, estuary and marine ecosystems).

A large scale study in the Melbourne region measured concentrations (at base flow and during storm events) of a number of nutrients and analysed their distribution in relation to TI (range: 0.1% to 49%) and EI (Hatt et al., 2004; Taylor et al., 2004). These studies only used catchments where land use was either urban or forested land and so removed confounding results that may have been driven by other land use such as industry, agriculture or horticulture. Median concentrations of total phosphorus (particulate and soluble) doubled and soluble phosphate quadrupled (~0.003 to 0.012

mg/L^{-1}) with increases in TI. Further analysis of the this data using step wise regressions indicated that soluble phosphate concentrations were best fitted to EI and that a value of 5% EI represented a break point where concentrations tended to stabilise (Walsh, Roy, et al., 2005). Nitrogen showed a different pattern with dissolved inorganic nitrogen (NO_3, NO_2 and NH_3 combined) and total nitrogen rising with septic tank density (0 to 141 septics/km^2) with highest septic densities between 4-12% TI and very few below 2% TI and above 30% TI as piped sewer systems became more common. Median dissolved inorganic nitrogen concentrations showed a 5 fold increase (0.3 to 1.8 mg/L^{-1}) with increased septic tank density, total nitrogen followed the same trend and doubled in concentration from ~0.8 to 2 mg/L^{-1}. Nearly the entire rise in total nitrogen and dissolved inorganic nitrogen concentration occurred in the range of 0-3.9% TI and 0-0.4% EI.

Although the concentration of nutrients is relevant to in-stream biological function (in particular algal or bacterial production) the sum of concentration and flow (defined as the load) determines the amount of nutrients delivered to downstream habitats. In the Melbourne study there was an increase in load per unit area of catchment as TI and IE increased. Loads of suspended solids, total phosphorus, total nitrogen, soluble phosphate and dissolved inorganic nitrogen increased by around 10 times as TI increased from 0.1 to 49% (Hatt et al., 2004). This data shows that although nutrient concentrations may drop under very high urban densities this may be a consequence of runoff increasing faster than the source of nutrients. An important implication of these results is that with decreased concentrations but higher efficiency of downstream transport nutrients are much less likely to be assimilated or processed in the stream leading to higher loads delivered to downstream water bodies.

5. Pollutants

Urban land use has long been associated with a range of pollutants in surface runoff (Weeks, 1982). Urban drainage from impervious areas has been shown to commonly contain a mixture of oil, grease, polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyl (PCB) and heavy metals (Allinson et al., 2014). Many of these pollutants are considered as toxicants but heavy metals and PAHs are of greatest concern because of their biological toxicity, persistence in the environment and potential for bio-accumulation. Another group of toxicants of emerging concern are micro-pollutants including pesticides, herbicides, hormones, pharmaceuticals and personal care products which can be biologically active at very low concentrations (Allinson et al., 2014). Many of the hydrological changes associated with urbanisation also increase the efficiency of delivery of these pollutants to streams and downstream receiving waters.

A final area of concern is the contamination of waterways with potential human pathogens sourced from urban infrastructure (primarily septic tanks but also domestic animals). Levels of *E. coli* are used as a tracer for warm blooded animal faecal contamination of water. In developing catchments septic tank density is considered the main potential risk of human faecal contamination. Additional factors that may determine the level of risk are the proximity of the septic tank to a waterway or the integrity and level of maintenance of the septic tank (Walsh & Kunapo, 2009).

6. Algal biomass and composition

As for nutrients benthic algal biomass increased by approximately tenfold (3 to 30 mg/m^2)with increasing TI and EI in the Melbourne study (Taylor et al., 2004). The increase in algal biomass was postulated to be primarily driven by release of filamentous green algae from phosphorus limitation through increased PO_4 concentrations in runoff (Taylor et al., 2004). Further analysis of this data indicated that maximum algal biomass was attained at between 2% and 5% EI depending on season (Walsh, Fletcher, et al., 2005).

Examination of benthic diatom species/taxa across the Melbourne urban gradient showed a clear distinction between sites above and below 1% EI in compositional structure (Newall & Walsh, 2005). European diatom derived indices of water quality showed a strong negative correlation with urbanisation indicating that diatom species/taxa composition was responding to degradation in general water quality (electrical conductivity, temperature, suspended sediments), similarly two other diatom indices designed to detect nutrient enrichment also showed a strong negative relationship with urbanisation (Newall & Walsh, 2005). Overall changes in both the biomass and composition of benthic algae was postulated to be driven by a combination of changes in salinity (measured as electrical conductivity median range across all sites 70-700 μS cm^{-1} with a break point in diatom composition at ~300 μS cm^{-1}) and increased supply of soluble phosphorus through frequent small flow storm events (Newall & Walsh, 2005; Taylor et al., 2004).

7. Macroinvertebates

Macroinvertebrates species have a central ecological role in many stream ecosystems and may be vital for the "health" of whole river networks (Clarke et al., 2008; Urrutiaguer, 2016). Many studies have shown a decrease in invertebrate diversity and abundance across urban gradients (Paul & Meyer, 2001) and this group of organisms has been considered as one of the most useful for comparing inter-regional responses to urban land use (Walsh, Roy, et al., 2005). In Australia the response of invertebrate communities to urban effects has been extensively used as surrogate for aquatic condition and in particular the SIGNAL score (Stream Invertebrate Grade Number –Average Level) has been used for many decades in the Melbourne region (Urrutiaguer, 2016). Typical responses of invertebrates to urban stress are a loss of taxa sensitive to disturbance and an increase of taxa typical of highly urbanised streams (Walsh et al., 2007).

Two studies of urban and forested land effects around Melbourne have shown rapid decreases in invertebrate diversity at very low levels of impervious cover, with very few sensitive species occurring at levels of TI of 4% in the Yarra River (Walsh et al., 2007) and 6-15% EI in small streams of the Melbourne region (Walsh et al., 2004). A more detailed study of both species and families of macro invertebrates from 572 sites across the Melbourne region (Walsh & Webb, 2016) used a more refined measure of effective impervious which weights the effect of the impervious area by the distance to the nearest stream or drain and is termed **Attenuated Impervious (AI)** (Walsh & Kunapo, 2009). Walsh and Webb (2016) showed a decline in 51 of the 60 families recorded with increasing AI, with 24 families showing a steep decline and their probability of occurrence reducing to near zero at AI values of 3%, three of these families were not found at AI values >1%. A further 6 families showed a steep decline to low or intermediate probability of occurrence at 3% AI. A comparison of the effect of AI on genera/species versus families (figure 2) showed a much greater impact on genera/species at AI levels above 2.5% with 11 out of 60 families (18%) never recorded at AI >2.5% compared to 296 of 477 (62%) of genera/species (Walsh & Webb, 2016). The sharp decline in the probability of occurrence in whole families of invertebrates at AI values of <1% suggest a lack of resistance to small levels of urban stormwater stress (Walsh & Webb, 2016) with the results indicating that the lowest level of AI that at which a decline in the SIGNAL score could be inferred was 0.1 to 0.3% (equivalent to 1000-3000m^2 of directly connected impervious area per km^2). A comparison of the effect of AI versus **Attenuated Forest Cover (AF)** showed that intact riparian forest can marginally reduce the impact of AI for a small number of families that are tolerant to some level of urban impact, indicating that retaining riparian buffers is only likely to have a small effect on family occurrence if urban-stormwater derived stress is not addressed (Walsh & Webb, 2016).

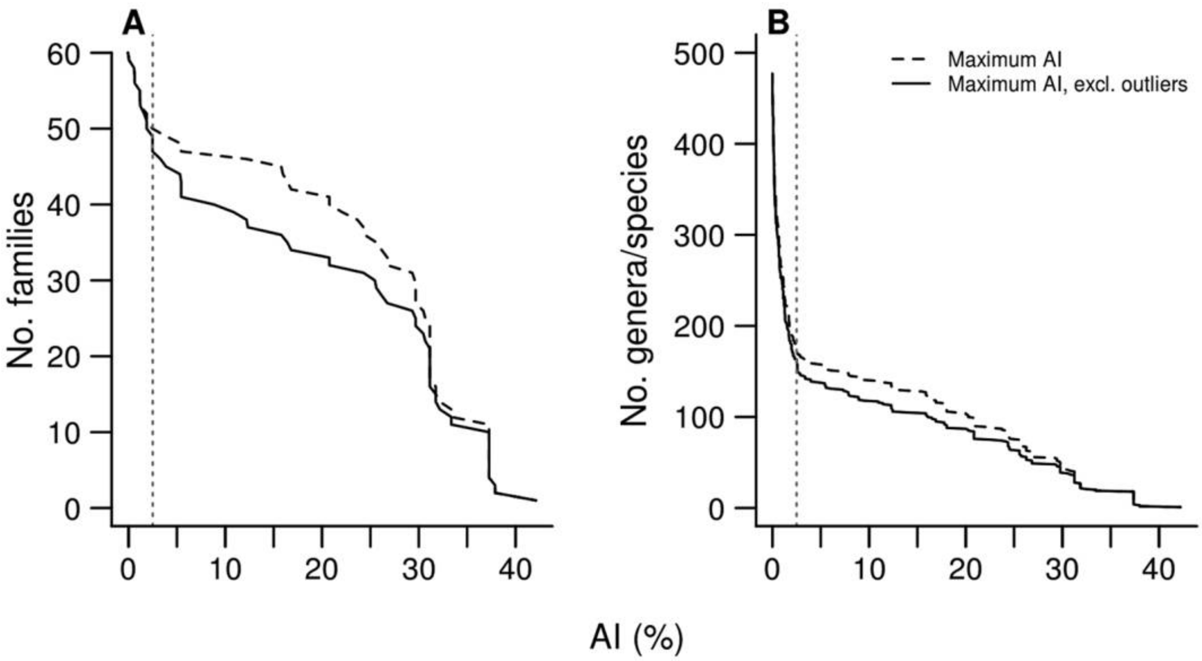

Figure 2. (Figure 7 of (Walsh & Webb, 2016)) Plots of the cumulative number (no.) of taxa that occur up to a particular value of attenuated imperviousness (AI) for family-level records (A) and the same records identified to genus or species (B). Data are for taxa recorded in the Melbourne region from the 60 families modeled in our study including data from additional locations (Fig. S1C). In each plot, taxon occurrences are ordered by the maximum AI value from which they have been recorded (maximum) and the maximum AI value ≤ 1.5× the interquartile range (maximum excluding [excl.] outliers). The plots show that most families were collected from streams with >2.5% AI (dotted vertical line), but that most genera/species were not recorded from streams with >2.5% AI.

8. Indicators of stream ecological condition

A number of water column and stream bed physical, chemical and biological indictors are commonly used to assess stream "health". Many of these indicators have been chosen due to their association with primary drivers to ecological degradation in running waters (Table 2). Increased values of abiotic indicators that typically increase with reductions in ecological values are; nitrate (NO_3), ammonia (NH_4), Total Nitrogen (TN), phosphate (PO_4), total phosphate (TP); dissolved organic carbon (DOC); total suspended solids (TSS); electrical conductivity (EC) and temperature (^oC). Increases in the water column concentration of all of the nutrients (NO_3, NH_4, TN, PO_4 and TP) as well as DOC and TSS generally lead to greater loads of these elements being delivered downstream waters.

Commonly used biotic indicators that often increase in association with decreased ecological function are algal biomass both in the water column and on the stream bed. More sophisticated biotic indicators of biological diversity are benthic algal species composition (Newall & Walsh, 2005) and the presence or absence of macroinvertebrates at the family and order level (Gooderham & Tsyrlin, 2002). All of these indicators have been shown to vary in response to ecological stress and in many cases indicator variables have been selected due to their high sensitivity to impacts of urbanisation (e.g. SIGNAL, the Stream Invertebrate Grade Number –Average Level) (Stewardson et al., 2010).

TABLE 2. The primary threats to streams and rivers. (Modified from (Allan & Ibañez Castillo, 2009).)

	Proximate causes	Abiotic effects	Biotic effects
Habitat alteration	Land-use change including deforestation, urban development	Loss of natural flow variability, altered habitat. Reduced habitat and substrate complexity, lower base flows Altered energy inputs, increased delivery of sediments and contaminants, flashy flows	Reduced dispersal and migration, changes to water quality and assemblage composition. Reduction in biological diversity favoring highly tolerant species. Changes in assemblage composition, altered trophic dynamics, can facilitate invasions
Invasive species	Aquaculture, sports fishing, pet trade, ornamental plants	Some invasive species modify habitat, otherwise minor	Declines in native biota, biotic homogenization, can result in strong ecosystem-level effects
Contaminants	Nutrient enrichment from agriculture, municipal wastes, urban deposition, atmospheric deposition, waste disposal, organic toxins.	Increased N and P, altered nutrient ratios. Reduced pH. Increased trace metal concentrations (e.g., Hg, Cu, Zn, Pb, Cd). Organic toxins Increased levels of PCB, endocrine disruptors, some pesticides	Increased productivity, algal blooms, altered assemblage composition Physiological and food chain effects Toxic effects through biomagnification Physiological and toxic effects

At higher trophic levels indicators such as the ratio of the sensitive coho salmon to the more tolerant cutthroat trout have been used as indicators of urban stress with in the USA (Kennen et al., 2005; National Research Council, 2009). Similarly the likelihood of encountering male, female or immature platypus in the Melbourne region has been used to indicate urban stress (Martin et al., 2014).

In the USA the Index of Biological Integrity (IBI) is a integrated quantitative measure that has be used to distinguish among a range of aquatic conditions (poor through excellent). It uses a range of data including invertebrate species richness and composition, trophic composition, and fish abundance and condition but also incorporates professional judgment based on the relative sensitivity of each of these parameters to stressors (National Research Council, 2009). IBI indices have been developed for a number of USA states and are used to detect the effect of non point source stressors to ecosystems that may not be detected by reliance on water quality or a more limited biological indicator alone (Kennen et al., 2005). Figure 1 shows the significant relationship (P <0.0001) between the North Carolina IBI and percent urban land use.

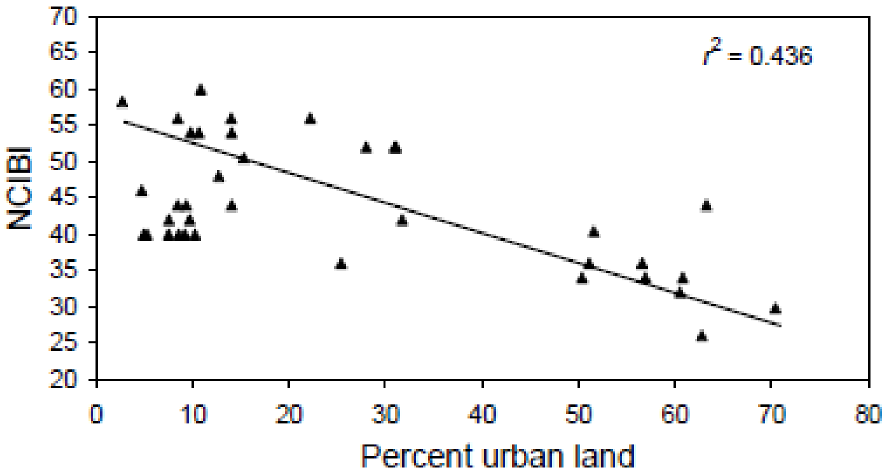

Figure 1. (from (Kennen et al., 2005)) Regression relation between percent urban land and the North Carolina index of biotic integrity (NCIBI).

9. Summary of impacts on steam ecological function of low urban density

Studies in Australia have shown that biological indicators (algal biomass, macroinvertebrate biodiversity and platypus numbers) show steep declines from 0% to <10% TI. Similarly A broad scale study in Connecticut showed that all catchments with TI >12% failed a macro invertebrate index for stream health (Figure 3). Results from the Connecticut study clearly show the high level of variability in stream ecosystem response to TI at low levels of imperviousness. Most streams in the range of 5-12% TI failed the macroinvertebrate index and a substantial proportion of streams at 2-3% TI also had very low scores (Figure 3). All streams with greater than 12% TI failed the index of stream health (Coles, 2012).

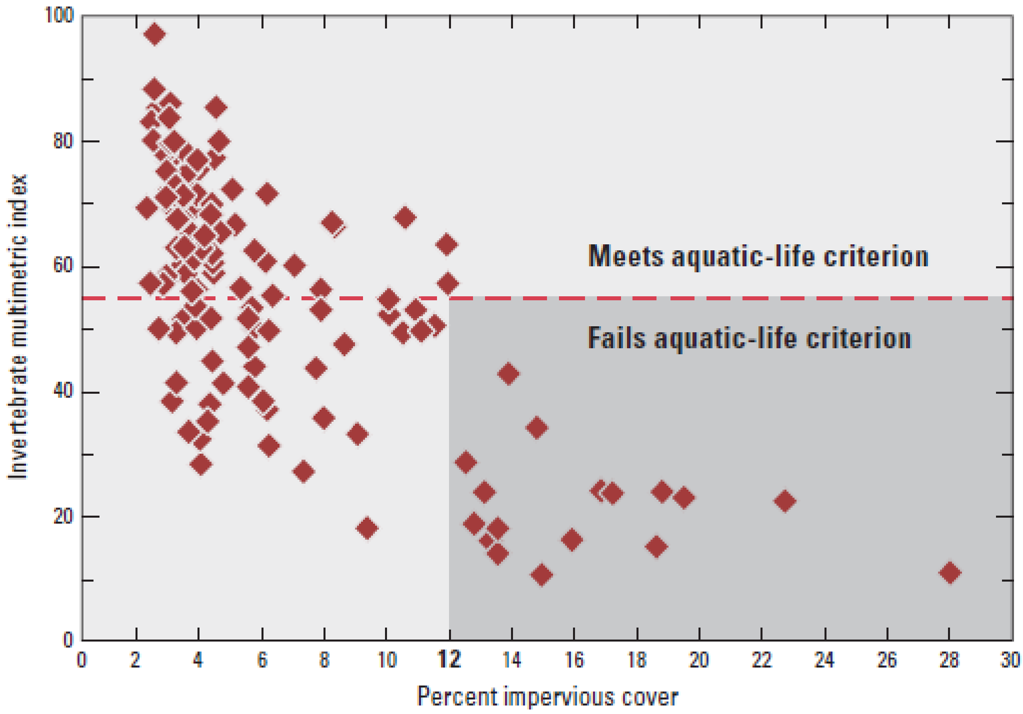

Figure 3. (Figure 7-1 of (Coles, 2012)) The Eagleville Brook impervious cover TMDL(Total Maximum Daily Load) is based on a Connecticut Department of Environmental Protection study that indicated streams in watersheds with impervious cover exceeding approximately 12 percent (the darker area) failed to met the Connecticut aquatic-life criterion for healthy streams.

There is a growing body of literature that has studied the impacts of urbanisation on abiotic and biotic components of steam function. A consistent result of these studies is that stream quality begins to decline from the lowest level of urbanisation measurable by current land use data (Walsh & Webb, 2016) and that degradation of aquatic biological communities begins at the onset of urban development (Coles, 2012). The extent which ecological function is compromised at low levels of urbanisation is not always clear as biological indices of steam health are often designed to detect changes in the occurrence of species known to be sensitive urban stressors. The rapid decline of organisms higher in the food chain (such as platypus) to very low levels of imperviousness (<3%) indicates a substantial change in ecological function. The data shows that macroinvertebrate biodiversity at both the stream reach and catchment level can be severely impacted at very low levels of urban density with macroinvertebrate species richness rapidly declining between 0% and 2.5% AI (King et al., 2011; Walsh & Webb, 2016).

A consistent impact of urbanisation is increases in concentrations of soluble and particulate nitrogen and phosphorus which are detectable at low levels of urbanisation (<2% EI) which are implicated in changed nutrient processing rates in the stream and increased algal biomass. Increased depositional nutrients delivered from impervious surfaces are almost always associated with increased contaminant loads, with many of these contaminants having not been assessed for their aquatic toxicity as they are relatively novel compounds. A study in Melbourne of eight urban sites sampled on two occasions detected 14 metals with copper and zinc found in all samples, in addition 15 herbicides and 93 semi-volatile organic chemicals were found in at least one sample (Allinson et al., 2014). This study also tested all samples against a toxicity bio-assay using bacteria and algae and found that all samples were moderately or strongly toxic to bacteria and all but two sites were toxic to microalgae (Allinson et al., 2014). The close association of a new suite of toxicants with the more commonly assessed nutrients, sediments, pesticides, metals and physicochemical changes in water quality has not been assessed at low levels of urban impact; however they remain a potentially important stressor to the biotic integrity of streams and receiving waters at very low levels of concentration.

It is still unclear which stressors cause the declines in stream biota observed at low levels of urbanisation. It is quite probable that different stressors may be more important under different catchment conditions and with different types of urbanisation (townships, clustered versus diffuse development). There are a number of commonly measured stressors that can be directly related to changes in biota such as nutrient enrichment leading to increased algal biomass; salinity and toxic metals impacting bacterial, algal or macroinvertebrate survival; or sediment smothering invertebrates or fish gills. Many of these stressors frequently increase together; hence the influence of one factor is often difficult to distinguish from a suite of potential impacts. Similarly there may also be a synergistic effect of multiple stressors or toxicants that lead to a greater impact than would be predicted from each stressor individually.

10. Threats to ecologically sensitive waters

Loads of nitrogen, phosphorus and sediments generated from urban areas delivered to downstream waters shown a linear increase with increasing urbanisation. Increases in upper watershed catchment urbanisation are almost always going to lead to increased loads of nutrients and sediments to slower flowing water bodies (reservoirs, lakes, low land rivers, coastal waters and estuaries). The magnitude of the increased loads will be determined by the level of urbanisation, proximity to watercourses, direct connection of impervious areas, climate, topography, vegetation cover and geomorphology (soils types). Increased loads of both nutrients and sediments to estuaries have been a primary concern for the ecological health of these systems. In particular smaller estuaries are more susceptible to eutrophication due to their low buffering capacity and limited nutrient processing and assimilation rates. This is particularly the case in intermittently open or permanently closed estuaries or coastal lagoons.

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