WETLAND TRACKER

GREAT BARRIER REEF CATCHMENT WETLAND CONDITION MONITORING PROGRAM

CONSTRUCT VALIDITY OF WETLAND TRACKER INDICATORS – A LITERATURE REVIEW

Wetland Tracker

Great Barrier Reef catchment wetland condition monitoring program

Construct validity of Wetland Tracker indicators – a literature review

September 2023

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The Department of Environment and Science acknowledges Aboriginal peoples and Torres Strait Islander peoples as the Traditional Owners and custodians of the land. We recognise their connection to land, sea and community, and pay our respects to Elders past and present.

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Main cover image: Maria Vandergragt

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Acronyms

DEC	Department of Environment and Conservation (Western Australia)
DEHP	Department of Environment and Heritage Protection (Queensland)
DELWP	Department of Environment, Land, Water and Planning (Victoria)
DENR	Department of Environment and Natural Resources (Northern Territory)
DERM	Department of Environment and Resource Management (Queensland)
DES	Department of Environment and Science (Queensland)
DEWHA	Department of the Environment, Water, Heritage and the Arts
DPSIR	Driver Pressure State Impact Response
DSITIA	Department of Science Information Technology, Innovation and the Arts (Queensland)
GBR	Great Barrier Reef
GBRCA	Great Barrier Reef Catchment Area
IBI	Index of Biological Integrity
PC	Pressure Class
QLUMP	Queensland Land Use Mapping Program
RE	Regional Ecosystem
WEV	Wetland Environmental Value
WT	Wetland Tracker

Glossary

Allelopathy	Occurs when an organism produces biologically active chemicals that inhibit or prevent the germination, growth and reproduction of other organisms.
Alpha diversity	Within-sample diversity.
Anoxia	In freshwater systems, having no or very little dissolved oxygen in a water body.
Anthropogenic	Caused by human activity.
Areal	Relating to an area, that is, a bounded two dimensional space.
Attribute	In an ecosystem context, an attribute is a biological, physical or chemical characteristic or feature inherent to an ecosystem. Attributes are aspects of ecosystems that can be evaluated, such as water quality and ground cover.
Baseline	Starting point used for comparisons between assessments to detect a change or trend in wetland condition over time.
Cadastral	Related to land parcel boundaries defined through land surveying.
Catchment	A drainage basin. An area of land from which runoff collects to a specific zone, usually defined by a wet area such as a wetland, river, lagoon or bay.
Commuting	Of connectivity processes, the movement of organisms among habitat patches in the process of daily activity (as compared with dispersal or migration).
Conceptual framework	A generalised illustration of the expected cause and effect relationships among main variables of interest. DPSIR is a conceptual framework widely applied in socio-ecological analysis to characterise cause and effect relationships among drivers, pressures, states, impacts and responses as defined elsewhere in this glossary.
Conceptual model	An illustration of parts of a system and the relationships that link these parts, specifying how the parts interact. A specific realisation of a conceptual framework such as DPSIR.
Construct validity	The degree to which an indicator addresses the underlying theoretical construct it is supposed to measure.
Desktop	Desktop assessment methods use maps, aerial images and remotely sensed geographical information to assess the condition of wetlands without going into the field.
Diatoms	Single-celled algae with highly ornamented cell walls made of natural glass. These cell walls persist in sediments long after the biological components have disappeared, allowing diatoms to be used as paleolimnological indicators.
Driver	A driver is something that causes a process to start. In the Driver–Pressure–State–Impact–Response (DPSIR) conceptual framework, drivers are anthropogenic influences and/or natural conditions driving environmental

	change. Wetland Tracker takes land use as the primary driver when assessing wetland condition.
Eco-exergy	The *available* energy of an ecosystem. An integrated representation of the biomass and genetic information of an ecosystem's biochemical components.
Ecological trap	An ecological trap occurs when rapid environmental change leads an organism to prefer habitats where their fitness is lower than in other available habitats.
Ephemeral	Of wetlands, those that temporarily contain water after rain events and dry out periodically on seasonal or longer (multi-year) time scales.
Eutrophication	Excessive plant and algal growth in a water body due to increased availability of nutrients that are usually limited in the system.
Exotic plant	For Wetland Tracker assessments, exotic plants include any plants not indigenous to the area of interest, including cultivated crop and pasture species originating elsewhere, plus any plants listed in the current Census of the Queensland flora as: (a) Naturalised in QLD or (b) Naturalised for the pastoral district encompassing the area of interest.
Floodplain	Land adjacent to a waterway that is naturally subject to occasional or periodic flooding. Floodplains can be narrow, or wide and flat with steeper sides at the edges.
Flow	The volume of water moving past a particulalr point (e.g. in a wetland or channel) per unit of time. Flow can describe surface and/or ground water flow.
Flow regime	The characteristic pattern of variation in water flow over a period of time, described by flow magnitude, frequency, duration, timing, and rate of change.
Fluvial	Stream- or river-related processes. Pertaining to flow.
Geodatabase	A collection of geographic datasets of various types held in a common file system folder.
Geomorphology	The study of landforms and how they develop.
Hazard	In ecological assessment, a hazard is a *source* of potential harm to the system. In contrast, a *risk* is the likelihood that harm will occur.
Holocene	The last ≈ 11,700 years of the Earth's history since the end of the last major ice age. The current geological epoch.
Hydroperiod	The seasonal pattern of water level in a wetland.
Hypoxia	In freshwater systems, low or depleted dissolved oxygen in a water body.
Impact	In a DPSIR framework, impacts are biological economic and social effects of environmental change.
Index	A compound measure that aggregates multiple indicators. Also sub-index.

Indicator	A measurable entity or process whose existence in an area is strongly correlated with specific environmental conditions that are desired to be measured.
Integrity	The ability of a system to maintain its organisation in the face of changing environmental conditions.
Lacustrine	Lake-like; referring to large, open, water-dominated systems.
Landscape scale	In a three-tier ecosystem investigation, landscape scale is a tier one investigation conducted using maps, remote imagery and GIS.
Littoral zone	The littoral zone of a wetland is the nearshore area between the terrestrial ecosystem and the deeper water of the wetland. In healthy wetlands it is characterised by the penetration of enough light to let aquatic plants flourish.
Macrobenthos	Organisms visible to the naked eye that live on the substrate (e.g. sand, mud) of water bodies.
Macroinvertebrates	Aquatic animals without backbones that can be seen with the naked eye.
Macrophyte	An aquatic plant that can be seen with the naked eye.
Micro-topography	Structured variability in soil elevation, both vertical relief and surface roughness, at about the scale of individual plants.
Monospecific	Relating to or consisting of only one species.
Nutrient	Chemical substances like nitrogen, phosphorus, carbon, that organisms need to effectively grow, survive and decompose.
Paleolimnology	Uses physical, chemical and biological indicators preserved in sediment to reconstruct past environmental conditions of aquatic systems.
Palustrine	Swamp-like; primarily vegetated, non-channel environments.
Pest plant	A weed; an exotic plant, including any plant not indigenous to the area of interest, that reduces the overall quality and function of a natural wetland ecosystem. Pest plants can include cultivated crop and pasture species.
Photosystem II herbicides	Substances that interfere with the transport of electrons needed for photosynthesis by binding to a specific protein (D1) and blocking the completion of natural processes within a plant.
Phytoplankton	Microscopic algae.
Ponded pasture	Wet areas of pasture established by creating earth walls (bunds) to contain freshwater and surface runoff.
Pressure	A pressure arises from a driver (e.g. human activities like land use) and directly affects the environment.
Propagules	Any material, such as seeds, spores or eggs, that carry an organism to the next stage of its life-cycle.

Rapid assessment	An assessment giving a broad view of a subject at a particular time. A rapid assessment is conducted in the shortest time frame that will produce reliable and valid results for its intended purpose.
Response	In a DPSIR framework, responses are actions of governments and societies in the face of environmental change. In DPSIR, a response may be targeted at drivers, pressures or impacts.
Risk	In ecological assessment, a risk is the likelihood that harm will occur. In contrast, a *hazard* is a source of potential harm to the system.
Scopus	An bibliographic database produced by Elsevier that enables online literature searches.
State	The condition of an ecosystem and its components in a certain area at a specific time that can be quantitatively and/or qualitatively described based on physical, biological and/or chemical characteristics.
Temporal	Relating to time.
Trophic	About eating and nutrition.
Vegetation succession	The process by which species composition or structure in an area changes through time.
Water regime	The prevailing pattern of water presence and inundation over a period of time. Its features include timing, duration, frequency, depth, extent and variability.
Watershed	The area of high ground that defines which water entering the landscape (e.g. through precipitation) flows into a river or wetland.
Wetland buffer zone	The transition zone between the wetland and the surrounding land use. Well-managed buffers support the ecological functions and values of wetlands (DERM 2011).
Wetland condition	Ecosystem condition is the overall quality of an ecosystem asset (United Nations et al., 2012). Wetland Tracker assesses two aspects of wetland quality within a Driver–Pressure–State–Impact–Response conceptual framework – the amount of anthropogenic pressure on a wetland and the state of its environmental values.
Wetland Environmental Values	Wetland Environmental Values (WEVs) are based on the physical and biological characteristics associated with a particular wetland. WEVs *support* the wetland's ecological processes and *underpin* its ecological, social and economic benefits. These benefits are sometimes referred to as ecosystem goods and services.
Wetland Tracker	A rapid assessment method for assessing the condition of freshwater floodplain wetlands in the Great Barrier Reef Catchment Area.
Xeric	Containing little moisture. Very dry.

Introduction

Background

Wetland Tracker (WT) is a rapid assessment method for assessing the condition of freshwater floodplain wetlands in the Great Barrier Reef Catchment Area (GBRCA). Wetland condition is the overarching term used to describe the pressures on, and the state of, wetland environmental values (WEVs). Change in wetland condition is a measured reduction or increase in pressures on wetlands and/or shifts in state showing an improvement or decline in the environmental values of natural wetlands within the monitored population (Tilden and Vandergragt 2022).

The conceptual model for WT is based on theorised causal links between anthropogenic drivers, anthropogenic pressure on freshwater wetlands, and the state of WEVs, underpinned by a Driver–Pressure–State–Impact–Response (DPSIR) framework (EEA 1999). DPSIR was developed to provide an effective causal framework for describing, assessing and reporting on the interactions between society and the environment (DSITIA 2015). Drivers (social, economic or environmental) apply Pressures on a particular environment, which changes State as a result, leading to an Impact (social, economic or environmental) that may elicit a societal Response (which may then feed back to Drivers, Pressures, States or Impacts) (Smeets and Weterings 1999).

Using the DPSIR framework, the GBRCA WT wetland condition monitoring focus is on Pressures and State. Wetland condition is assessed using a set of 23 direct and indirect indicators informing eight sub-indices and two overall indices that represent either land-use driven *pressure* on wetlands or the *state* of WEVs (Figure 1).

For the purpose of WT, an indicator is defined as a measurable component or process whose existence is strongly correlated with specific environmental conditions that are to be measured. Pressure indicators are defined as indicators of pressures, not the actual pressures. State indicators are defined as indicators of states, not the actual states. The use of such correlated indicators (surrogates) is an inevitable feature of rapid assessment instruments.

Several of the WT indicators incorporate or are based on measures of vegetation. This is because wetlands in the GBRCA vary considerably in their cycles of wet and dry phases, and measures of vegetation, unlike direct indicators of water quality or wetland faunal populations, can be assessed regardless of whether wetlands are wet or dry (Tilden and Vandergragt 2022). Furthermore, single community types, such as plant communities, can act as surrogates for other communities and are commonly used to provide information on overall ecosystem condition (Rooney and Bayley 2012; Kutcher and Forrester 2018).

Four classes of anthropogenic pressure anchor the WT conceptual framework (DSITIA 2015). The pressure classes, each focussing on direct pressures that can be managed by land holders and comprising two or more individual indicators of anthropogenic pressure on wetlands, are:

- PC1: Biological (pest) introductions (e.g. plant pests and animals changing the wetland)
- PC2: Habitat modification (e.g. loss of natural vegetation around the wetland)
- PC3: Change to water regime (e.g. natural wetland water levels altered by a dam or levee)
- PC4: Input pressures (e.g. chemicals and nutrients going into the wetland).

The four wetland environmental values, each comprising two or more individual indicators of the state of wetland condition, are:

- WEV 1: The biological health and diversity of the wetland's ecosystems (biotic integrity)
- WEV 2: The wetland's physical state and integrity (local physical integrity)
- WEV 3: The wetland's natural hydrological cycle (local hydrology)
- WEV 4: The natural interaction of the wetland with other ecosystems, including other wetlands (connectivity).

Consistent with the DPSIR framework, an implied causal link between *pressure* and *state* is supported by a high correlation between WT overall pressure and overall state indices (Australian and Queensland Governments 2016; 2019). Theoretically, the state of a wetland may deteriorate in response to anthropogenically generated pressure on its environmental values or improve as these pressures are managed and mitigated. Causal links are also implied among individual pressure and state indicators, again consistent with DPSIR. At this stage in the development of WT, causal links between *particular* pressures and their correlated state variables remain to be explored through detailed studies.

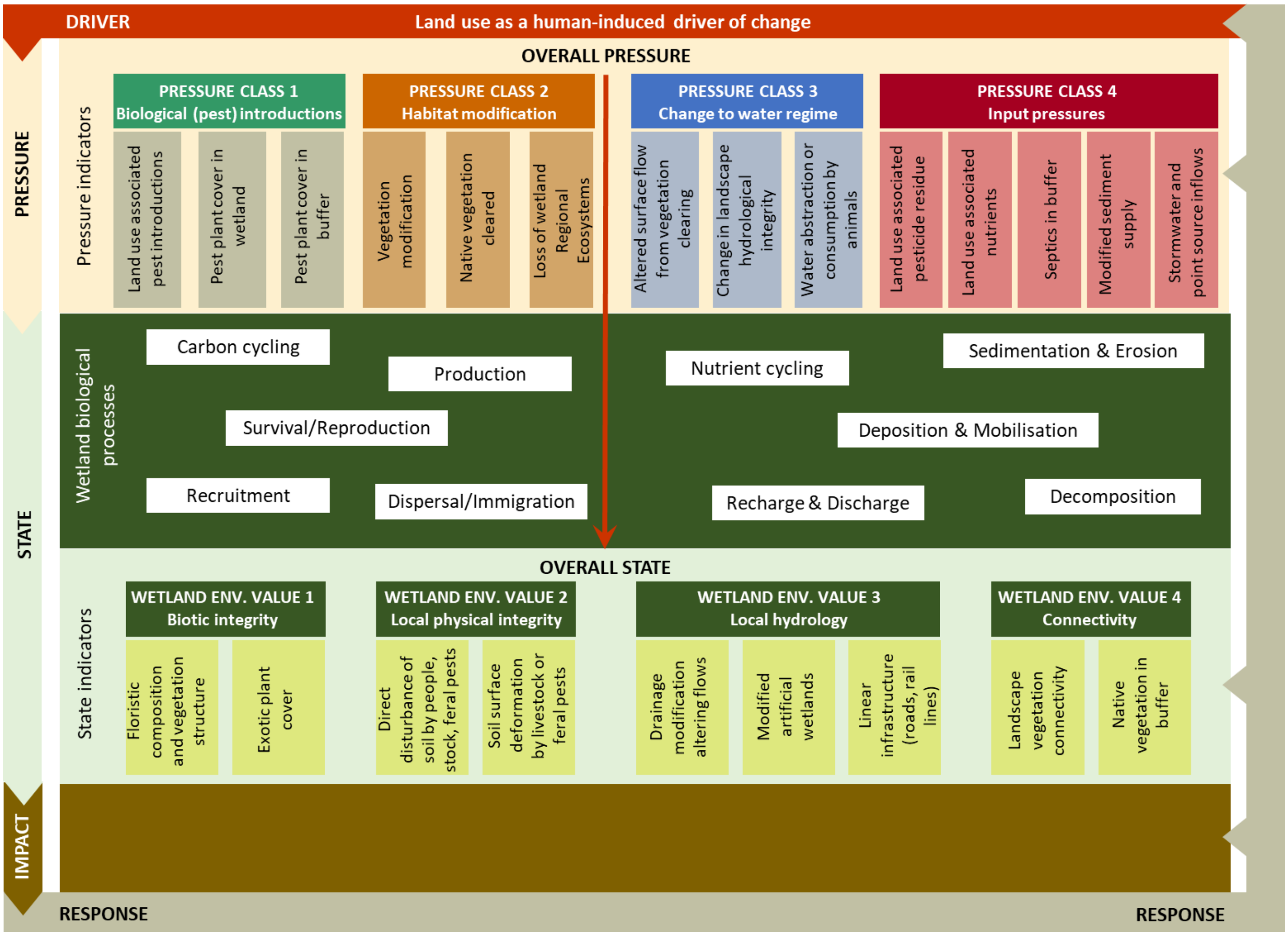

Figure 1 Overarching Driver–Pressure–State–Impact–Response conceptual model for Wetland Tracker. Specific conceptual models have been developed for each pressure subindex (Pressure Class 1-4). The causal links between each pressure and the state subindices (Wetland Environmental (Values 1-4) with which they are correlated remain to be explored through detailed studies. ENV., Environmental.

Aim and scope

WT was informed in its development by guidelines for evaluating ecological indicators (Jackson et al. 2000) under the main headings of conceptual relevance, feasibility of implementation, response variability, interpretation and utility. Under the heading of conceptual relevance, the guideline states that:

> *It must be demonstrated that the proposed indicator is conceptually linked to the ecological function of concern. A straightforward link may require only a brief explanation. If the link is indirect or if the indicator itself is particularly complex, ecological relevance should be clarified with a description, or conceptual model*

> *… This information should be supported by available environmental, ecological and resource management literature.* (Jackson et al. 2000)

As such, the aim of this review is to provide an evidence base (rather than a systematically comprehensive review), drawn from available environmental, ecological and resource management literature, to support the conceptual links between WT pressure and state indices and indicators, and the condition of freshwater wetlands. In other words, this literature review aims to support the construct validity of the WT indices and indicators.

Relationships among the WT indicators of land-use driven pressure on wetlands and the state of WEVs are conceptual, underpinned by expert opinion (DSITIA 2015) and literature exploring the impact of land use on freshwater wetland condition. *Wetland Tracker Part 4* (this document) presents a review of that literature, gathered using a combination of systematic and more traditional, narrative-style review methods. The methods are outlined below, followed by the review proper, first for each pressure class subindex and its component pressure indicators, and then for each WEV subindex and its component state indicators.

Literature search methods

To identify literature that explored and thus underpins conceptual relationships between the WT indices and indicators of land use pressure on wetlands and the state of WEVs, a set of key literature searches were conducted in Scopus in August 2019. These searches focused on the WT pressure-class indices and indicators, under the assumption that literature exploring such pressures would also cover material on their relationships with wetland condition (and therefore WEV indices and indicators). The searches used search strings relevant to each of the four pressure classes, using operators and symbols (e.g. * and ? for character truncation), as follows:

For PC1 Biological (pest) introductions

> (wetland* AND (condition OR state OR health OR impact) AND ("introduced species" OR introd* OR pest OR exotic OR weed OR pig OR cattle))

For PC2 Habitat modification

> wetland* AND (condition OR health OR state OR impact) AND ("riparian vegetation" OR buffer OR "vegetation clearing" OR "wetland regional ecosystem*" OR (vegetat* AND (modif* OR clear*)))

For PC3 Change to water regime

> wetland* AND (condition OR health OR state OR impact) AND (hydrolog* OR "water regime")

For PC4 Input pressures and the main classes of potential wetland pollutants

> **Pesticides**
>
> wetland* AND (condition OR health OR state OR impact) AND (pesticide* OR herbicide*)
>
> **Nutrients**
>
> wetland* AND (condition OR health OR state OR impact) AND (nutrient* OR fertili?er*)
>
> **Sediment**
>
> wetland* AND (condition OR health OR state OR impact) AND sediment*
>
> wetland* AND (condition OR health OR state OR impact) AND erosion*
>
> **Stormwater**
>
> wetland* AND (condition OR health OR state OR impact) AND (stormwater* OR "point source")

Literature published earlier than 1995, in languages other than English or in non-environmental, non-ecological or non-natural resource management subject areas (e.g. Maths, Psychology) were excluded. Literature was also excluded based on keywords that suggested the publications would relate to non-natural or non-freshwater wetlands (e.g. 'constructed wetlands'). Relevance of the publications that resulted from each search was further established by scanning the publications' abstracts.

Once the relevance of publications was established, the literature was reviewed in relation to each pressure class and each WEV, keeping the following questions and the DPSIR framework in mind:

- Does the publication identify one or more negative or positive impacts of land use on wetland condition?
- To what pressure is the impact attributed?
- To which WT indicator(s) is the pressure relevant (overall pressure, specific pressure classes and/or individual pressure indicators)?
- What is being impacted (overall state, specific WEVs and/or individual state indicators)?

Where no or very few references from the above search strings could be linked to a specific pressure class or WEV index or indicator, additional searches were conducted focusing on just that index or indicator. For example, one such search for the state indicator *S13: Landscape connectivity* was:

wetland* AND (vegetat* OR forest) AND (connect* OR corridor OR belt OR remnant OR patch)

Searches of this sort were conducted for the following indicators:

- Number of septic systems within 200m of the wetland, per ha of mapped wetland (P5)
- Drainage modification and artificial structures altering natural surface flows (S9)
- Landscape vegetation connectivity (S13)
- Native vegetation in the 200m buffer (S14).

Additional searches were also conducted to further elucidate the links between WEVs and their individual state indicators, using search terms relating to the biotic integrity, local physical integrity, local hydrology, and connectivity of wetlands.

Finally, the reviewed body of literature supporting WT conceptual modelling and the construct validity of the WT state indices and indicators was broadened to include:

- Key references, known to the WT development team through their expertise in wetland ecology (sometimes this was work published before 1995)
- Relevant references identified from the reference lists of publications resulting from the Scopus searches (again, often published before 1995)
- New research brought to the attention of the team through their regular monitoring of the domain of wetland science literature.

Indicator links to wetland condition: pressure index

Pressure Class (PC) 1 – Biological (pest) introductions

PC1 comprises three individual indicators, one that indirectly assesses the presence of pest plants and animals (P1) by surveying wetlands for surrounding land uses associated with the introduction of exotics species known to disturb wetlands (DSITIA 2015) and two that directly assess the aquatic and terrestrial pest plant cover within wetlands (P7) and their 200 m buffer zones (P8).

Introduced exotic plants and animals have been found to have a range of impacts on freshwater wetlands and their buffer areas in tropical and subtropical Australia. In a review of threats and management options, Adame et al. (2019) summarised the major impacts of invasive species in the GBRCA: invasive non-native macrophytes reduce oxygen levels in the water column (through several processes), which in turn affects fish and invertebrates (including through hypoxia events), while non-native aquatic and terrestrial vertebrates outcompete native species and introduce diseases.

In terms of plants, introduced macrophytes can change water chemistry and cause poor water quality, for example via their decomposition or by completely covering the surface of waterbodies thereby inhibiting re-oxygenation of the water column (Perna et al. 2012a). Introduced plants can also compete with native species, altering the structure of plant communities (Wurm 2007; Catford et al. 2011; Price et al. 2011), and reduce habitat suitability for native vertebrates (fish, amphibians, water birds) (Houston and Duivenvoorden 2002; Parris and Lindenmayer 2004; Perna et al. 2012a; Arthington et al. 2015) and macroinvertebrates (Houston and Duivenvoorden 2002; Douglas and O'Connor 2003).

In terms of introduced aquatic animals, exotic fish can compete with native fish species, prey upon native fish, amphibians and invertebrates (Mazumder et al. 2012; Hamer and Parris 2013), alter water quality and degrade aquatic habitat (Stuart et al. 2021). Introduced turtle species can out-compete native turtles for food, impact waterbird nesting and basking areas, and generally damage habitat (State of Queensland 2016).

Introduced terrestrial animals such as pigs and cattle can have major biotic and physical impacts on freshwater wetlands. Pigs and cattle feed on and dig up native vegetation, increase nutrient concentrations, resuspend sediments and modify macrophyte communities, with the resulting pH changes and anoxia affecting the habitat of many animals (Adame et al. 2019). For example, fish can be exposed to risk of overheating and asphyxia as a result (Waltham and Schaffer 2018). In tropical wetlands, feral pigs can change the physical structure of wetlands, making them shallow and uniform in profile, and removing areas that could otherwise protect fish from high water temperature. Doupé et al. (2010) report that in north Queensland wetlands, pig presence can alter vegetation assemblages, increase the proportion of open water and bare ground, and reduce water quality. Marshall et al. (2020) studied the impact of feral pigs on ground surface invertebrates in exposed wetland sediments and found pig damage limited both invertebrate richness and abundance. Pigs also affect the food web, feeding directly on native animals and providing food for crocodiles (Adame et al. 2019). Cattle can also affect wetland water and soil quality, and plant communities (Pettit et al. 2012). Lunt et al. (2012) found that livestock grazing (as a land use) favoured exotic annual plants over native perennials, especially if changes in flood regimes led to drying conditions.

More generally, the impacts of introduced species can interact with other classes of pressures, especially altered hydrology (Catford et al. 2011; Price et al. 2011; Mazumder et al. 2012; Lunt et al. 2012), which is captured by the PC3 index of WT. Introduced species can also interact with each other in their impacts on wetlands (Perna et al. 2012a; Lunt et al. 2012). This means that not only is the biotic integrity of wetlands disturbed by introduced species, but also, a wetland's physical integrity, hydrology and aquatic connectivity can be impacted negatively.

Indicator P1 – Land use associated with the introduction or perpetuation of pest species

Indicator P1 contributes to PC1 and assesses land use associated with the introduction or perpetuation of pest species within the mapped wetland and its 1 km buffer (a surrogate for a wetland's localised watershed).

Eighty-seven percent of the land area within in the GBRCA is modified or cultivated for grazing and dairy, sugarcane, horticulture and mining (Adame et al. 2019, Arthington, et al. 2018). The link between different types of land use and the introduction of exotic species detrimental to the condition of wetlands is well established. Adame et al. (2019) reviewed threats to wetlands in the GBR catchments and their management, identifying land use associated factors such as nutrient runoff and hydrological alteration as favouring the establishment of invasive macrophytes. As highlighted above, such plants can reduce oxygen levels in the water column, in turn negatively impacting native fish and invertebrates. Pest plant species in the wetland and buffer zone also have the capacity to act as vectors for weed dispersal and as pathways through the landscape for terrestrial animal pests (DSITIA 2015). Invasive fish are spread through irrigation networks associated with cropping (Russell et al. 2003). Livestock grazing favours exotic annual plants over native perennials (Lunt et al. 2012), while declines in the ecological condition of riparian habitats and loss of biodiversity of birds, frogs and plants have been associated with cattle grazing intensity in river red gum habitats (Jansen and Robertson, 2005). Quinn et al. (2011) found urban habitats in downstream parts of catchments were more susceptible to invasion by alien aquatic plant species than natural habitats. Parris and Lindenmayer (2004) established a link between exotic pine plantations surrounding freshwater ecosystems and reduced frog biodiversity. Pigs need water and food resources, as well as protection from heat stress and disturbance. This leads them to favour well-connected freshwater wetlands with suitable habitat (Froese et al. 2015) which are generally associated with less intensive agricultural uses and nature conservation (Australian and Queensland Governments 2016)

Indicator P7 – Plant pest cover in the mapped wetland

Indicator P7 contributes to PC1 and assesses the percentage cover of pest plant species in wetlands. Wetlands are vulnerable to pest plant invasions due to their landscape position, receiving inputs of nutrients, sediment, water and plant propagules from upland watershed areas and connected waterways (Zedler and Kercher 2004; De Rouw et al. 2018). Aquatic plant pests in wetlands have a range of impacts on wetland biology and hydrology. Compared with native wetland species or non-invasive exotics, different species of wetland pest plants have been found in numerous studies to outcompete and out-survive native plants (Rea and Storrs 1999; Hastwell et al. 2008; Price et al. 2011), suppress germination in native taxa (Wurm 2007), inhibit submerged plant growth and change water chemistry (Rea and Storrs 1999). Invasive aquatic plants in wetlands also alter animal habitat, leading to such impacts as reduced species richness in fish (Perna et al. 2012a; Arthington et al. 2015), in benthic and epiphytic macroinvertebrates (Douglas and O'Connor 2003) and potentially in waterbirds (Houston and Duivervoorden 2002), as well as reduced abundance of benthic invertebrates (Douglas and O'Connor 2003) and freshwater fish (Houston and Duivenvoorden 2002). On the other hand, by changing vegetation structure, pest plants may lead to an increase the abundance pest animals, for example exotic fish (Houston and Duivervoorden 2002; Perna et al. 2012a).

Pest plants can alter the hydrology of wetlands they invade, for example by changing flow patterns and altering freshwater drainage from floodplains (Rea and Storrs 1999). Of particular concern in the GBRCA are ponded pasture grasses such as olive hymenachne (*Hymenachne amplexicaulis*), para grass (*Brachiaria mutica*) and aleman grass (*Echinochloa polystachya*) that outcompete native plants and reduce the complexity of habitats, leading to a general reduction in faunal biodiversity. These introduced invaders also increase fuel loads resulting in hotter fires that in turn favour regrowth of exotic ponded pasture over native species (Rea and Storrs 1999; Houston and Duivenvoorden 2002; Douglas and O'Connor 2003; Wurm 2007; Arthington et al. 2015).

Water, nutrients and sediments act as catalysts for plant growth, while flood disturbance often creates canopy gaps for plants to colonise. Areas subject to frequent disturbance (e.g. flood scouring or cycles of inundation and drawdown) are readily colonised by weedy terrestrial species (Zedler and Kercher 2004). Many wetland and floodplain pest plants have traits that allow them to outcompete native species to colonise bare areas left after waters recede (e.g. high germination and growth rates, allelopathy, drought resistance), often forming dense monospecific stands (Zedler and Kercher 2004; Company et al. 2019; Gioria et al. 2019). Pest plants of wetlands, floodplains and riparian zones often have water- or wind-dispersed propagules for spreading rapidly over long distances (Nilsson et al. 2010; De Rouw et al. 2018). Other traits such as high levels of seed production or long-lived seeds can lead to large seed banks in wetland and floodplain sediments (Gioria et al. 2019). Accumulation of a soil seed bank of pest plant species during drawdown or dry phases can encourage reinfestation of these areas along with associated waterways and riparian zones (D'Antonio and Meyerson 2002).

Pest species often increase productivity in wetlands. As noted above, the resulting dense growth and litter build-up from productive invasive species can increase fuel loads and fire risk, particularly during dry periods (Zedler and Kercher 2004). Increases in vegetation biomass from growth of dense monospecific pest plant stands can impact wetlands in other ways. Such dense pest plant infestations can alter sediment retention and discharge rates, leading to increased infilling and loss of habitat for aquatic species. They can hasten drying rates either by increasing evapotranspiration or through the effects of sediment capture/infilling. Food webs can be altered by an increase in the volume and a reduction in the variety of available carbon and nutrient inputs from vegetation sources. Habitat complexity and diversity can be reduced, resulting in loss of habitat niches for some native species (Rea and Storrs 1999; Zedler and Kercher 2004).

Indicator P8 – Plant pest cover in the 200m buffer

Indicator P8 contributes to PC1 and assesses the percentage cover of pest plant species in the 200 m buffer zones of wetlands, rather than in the wetlands themselves (which Indicator P7 assesses). Wetlands and their immediate buffers are highly interactive zones in complex ecological systems. This is especially the case for ephemeral wetlands on floodplains whose natural cycles of wetting and drying cause a predictable and characteristic succession of vegetation changes as wetlands dry, then once again fill during a wet phase (State of Queensland 2010). The condition of a wetland’s buffer vegetation is often a reliable indicator of pressures on and the condition of the wetland itself (Naiman and Décamps 1997; Pusey and Arthington 2003; Watkins et al. 2011; Waltham et al. 2019). Anthropogenic pressures impacting the buffer also impact the wetland, often in diverse ways.

Plant pest species introduced to buffer zones by human activity such as vehicle, cattle and other animal movements are often generalists, capable of occupying a wide range of habitats within the wetland complex. These can then invade and degrade adjacent wetlands, outcompeting native specialists and changing conditions for wetland fauna (Rea and Storrs 1999). Changed water regimes can also facilitate the movement of pest species from buffer to wetland (Catford et al. 2011). Heavy weed infestations or exotic plantations in the buffer can isolate wetlands, reducing their suitability for fauna that need to move between waterlogged or wet areas and their adjacent terrestrial environment (Parris and Lindenmayer 2004).

Pressure Class (PC) 2 – Habitat modification

PC2 assesses impacts of habitat modification at two spatial scales: the area within 200 m of the mapped wetland boundary (Indicator P2) and, more broadly, the area within 5 km of the mapped wetland boundary (indicators P20 and P21), termed the 200 m buffer and 5 km buffer, respectively. Within these buffer zones, the indicators focus on change in the extent of natural vegetation (i.e. terrestrial vegetation that is not wetland associated, terrestrial vegetation that is wetland associated, and aquatic vegetation) and change in the extent of natural wetland vegetation (i.e. terrestrial vegetation that is wetland associated, and aquatic vegetation), the latter acting as a surrogate for modification of hydrological and ecosystem connectivity.

Wetland buffers are defined as the transition zone between the wetland or riverine ecosystem and the surrounding land use. They help protect and support the functions and values of wetlands (DERM 2011). A natural wetland buffer helps to mitigate impacts on the wetland from human land use (Naiman and Décamps 1997; Arthington and Pusey 2003; Adame et al. 2019; Waltham et al. 2019). The proportion of healthy native vegetation surrounding a wetland is an ecologically important habitat variable that is rapid and cost effective to assess (Rooney et al. 2012). The condition of vegetation surrounding a wetland is also highly correlated with other dimensions of a wetland's natural condition for example, its biodiversity, microclimate and water quality (Naiman and Décamps 1997; Houston and Duivenvoorden 2002; Parris and Lindenmayer 2004; Burger et al. 2010; Rooney et al. 2012; Adame et al. 2019).

Habitat modification in the buffer zone of wetlands can affect wetlands through multiple pathways and mechanisms, operating at different spatial scales (Allan and Johnson 1997). Habitat modification for human land use may alter vegetation, topography, hydrology, water quality and climate, all with potential impacts on the condition of freshwater wetlands, as described further below.

Indicator P2 – Modification of native vegetation in the 200 m buffer

Indicator P2 contributes to PC2 and considers (a) the percentage of native vegetation cleared, or replaced by exotic plant infestations, within a wetland's 200 m buffer, and (b) the lineal percentage cleared on the buffer edge of the mapped wetland.

Wetland riparian vegetation not only provides habitat in buffer zones, but also performs many ecological processes supporting wetland condition, including: transforming nutrients and chemicals introduced by human and natural processes in the wetland's catchment (Karr and Schlosser 1978; Naiman and Décamps 1997; Burger et al. 2010; Waltham et al. 2019), filtering sediment runoff before it reaches the wetland (Cooper et al. 1987; Allan and Johnson 1997; Naiman and Décamps 1997; Waltham et al. 2019), controlling the wetland's temperature and light regime (Pusey and Arthington 2003; Waltham et al. 2019) and introducing organic matter that either can enter the wetland food-chain via a number of trophic pathways or provide habitat for aquatic biota (Hessen et al. 1990; Naiman and Décamps 1997; Robertson et al. 1999; Pusey and Arthington 2003; Watkins et al. 2011). These important functions are altered or impaired when natural vegetation is partly or fully cleared.

In Adame et al.'s (2019) review on managing threats and restoring wetlands in the GBRCA, actions were identified for addressing eleven major categories of threats. Managing or remediating buffer zones and riparian vegetation were seen as important strategies to address the greatest number of listed threats, including nutrient, sediment and pesticide pollution, impacts of cattle and horses on wetlands and climate change impacts such as increased temperature and increased frequency of intense tropical storms.

Attum et al. (2008) found that the area of forest within a wetland buffer was significantly related to the occurrence of rare reptiles, up to a buffer width of 250 m (but not 500 m or 1000 m). Rooney et al. (2012) attempted to model biotic integrity of wetlands based on remotely sensed land cover. Plant-based Index of Biotic Integrity (IBI) scores were predicted using land cover data extracted from seven nested landscape extents (100 m to 3000 m radii). Plant-based IBI scores were predicted by land cover data at all spatial extents considered but were best predicted using data from 100 m buffers. Rooney et al. (2012) also modelled bird-based IBI scores and found that a 500 m buffer was the best predictor for birds. All these buffer widths (250 m, 500 m) are of the same order of magnitude as the WT 200 m buffer width, the widest area that can practically be ground-truthed within the timeframe of a rapid assessment.

Indicator P20 – Native vegetation cleared within 5 km of the wetland

Indicator P20 contributes to PC2 and assesses the percentage of native vegetation historically and recently cleared within 5 km of wetlands. The clearing of wetland buffer zones on a scale of kilometres severs natural vegetation and landscape connections, interfering with the ability of wetlands to interact ecologically, but also potentially physico-chemically and hydrologically, with other ecosystems (Kingsford 2000; Joyal et al. 2001; Attum et al. 2008; Burger et al. 2010). For example, Burger et al. (2010) found that the type of broad-scale land use (impacted, transitional or remnant) surrounding restored riparian zones had different effects on their soil chemistry, especially carbon and nitrogen concentrations, and hypothesised that this would have flow-on effects on the water quality of adjacent aquatic environments.

In Queensland, broadscale clearing and fragmentation of native vegetation is typical in agricultural development. For example, Pert et al. (2010) report that since European development in the Tully-Murray catchment of the Wet Tropics region (part of the GBRCA), the area of floodplain vegetation has decreased by about 80 percent, while the riparian area of the catchment has decreased by about 60 percent. The cleared land is used primarily for growing sugarcane, bananas and cattle. Habitat clearing for such development is significantly associated with changes in species richness and abundance of wetland fauna, along with changes in regional hydrology and drainage (Pert et al. 2010; Pearson et al. 2013; Arthington et al. 2015).

Land cover data extracted from the area within a 3 km radius of wetlands has been found to significantly predict plant and bird based IBI scores (Rooney et al. 2012; noting that the 3 km radius was the widest considered). Lehtinen et al. (1999) found that amphibian species richness in depressional wetlands was reduced at sites with moderate to high habitat loss and fragmentation (due to the replacement of forest by urban development) at 500 m, 1000 m and 2500 m scales, while Steen et al. (2012) and Joyal et al. (2001) examined the terrestrial habitat requirements of freshwater turtles, finding that different species nested up to ≈1500 m from the nearest wetland. Parris and Lindemayer (2004) studied the impact of pine plantations on frog diversity on a regional scale. They found that establishing pine plantations within a mosaic of intact native forest and native forest remnants resulted in varying degrees of isolation of forest patches and wetlands. Across a gradient, a two-fold increase in amphibian species richness was found between sites in pines and those in the class of intact forest furthest from pines.

The above studies show the importance, to the biotic integrity of freshwater wetlands, of buffer areas with intact native vegetation at scales of kilometres, suggesting that within five kilometres of a wetland, the condition of the surrounding vegetation is a useful indicator of habitat modification pressure on a wetland's environmental values.

Indicator P21 – Loss of wetland Regional Ecosystems (REs) within 5 km of the wetland

Queensland RE mapping provides comprehensive regularly updated information describing expected vegetation communities, including community composition and structure, for wetlands and their surrounding buffer areas (Queensland Government 2019). Indicator P21 contributes to PC2 and assesses habitat modification pressure on the connectivity of freshwater wetlands by comparing the extent of wetland RE vegetation in the 5 km buffer zone with its pre-European occupation ('pre-clear') extent. Causes of wetland vegetation loss, and hence wetland loss, include clearing and draining for agriculture and urban development, and climate change (Kingsford 2000; Blann et al. 2009; Saunders et al. 2020).

Natural wetlands within a 5 km radius of each other interact biologically and often hydrologically. Within the GBRCA, many wetlands are connected as components of complex floodplain ecosystems. Most Australian floodplain wetlands undergo a wet and dry cycle (e.g. Kingsford 2000, Saintilan and Imgraben 2012). During dry phases, they may be aquatically isolated for varying periods, depending on floodplain hydro-geomorphology and associated filling and drying regimes that may be seasonal or even decadal. Many factors related to land use may alter the length of natural dry phases (Sheaves et al. 2014; State of Queensland 2011a). When wetlands within ecological complexes are lost, due to a variety of land management and development processes such as clearing, water diversion and drainage, the natural interaction among floodplain wetlands is reduced or lost. This impacts the biotic, physical and hydrological condition of the remaining wetlands (Joyal et al. 2001; Attum et al. 2008; Burger et al. 2010).

Once a wetland becomes isolated, whether by natural factors (e.g. progress of a long dry spell) or anthropogenic factors (e.g. terrestrialisation of downstream wetlands due to large-scale removal of water from the system), its biota and ecological processes can experience altered trajectories. With timely local freshwater connections, wetland fauna will be exchanged with other nearby wetlands as part of a natural cycle. Prolonged isolation may be tolerated depending on local processes (e.g. natural adaptations, water level, habitat quality, predators, pollutants). If a wetland is starved of water for an extended period, a permanent terrestrial ecosystem may establish (Kingsford 2000; Lunt et al. 2012).

Reduced connectivity among wetlands and between wetlands and their river systems has been associated with a range of ecosystem impacts (Leigh et al. 2010). Thoms et al. (2005) aimed to quantify the fragmentation of a large floodplain ecosystem by modelling temporal and spatial effects of water resources development on the exchange of dissolved organic carbon between floodplain patches. Their modelled data suggested that the amount of dissolved carbon in floodwater was more than halved by the water resources development. Arthington et al. (2015) found that fish assemblages in isolated water bodies varied, during the dry season, with flooding and connectivity history. In floodplain ecosystems, connectivity facilitates fish movement into floodplain wetlands for refuge from predators, spawning, feeding and growth (Karim et al. 2014). Attum et al. (2008) found proximity to other wetlands affected the distributions of common reptiles. Connectivity has also been identified as key to determining fish and macroinvertebrate assemblages in tropical wetlands (Karim et al. 2014). As flooding facilitates exchanges of water, sediments, nutrients and biota among river channels and their floodplain patches (wetlands), loss of water and, over time, loss of wetlands from the system will impact the functioning and integrity of these complex systems. Maintaining a wetland’s hydrological connection with the aquatic ecology of the surrounding landscape is vital to the condition of its environmental values. As such, the restoration of connectivity of waterholes, wetlands and floodplains is considered vital in managing the effects of climate change on wetlands in the GBRCA, including increases in temperature, rainfall variation and more intense tropical storms (Adame et al. 2019).

Pressure Class (PC) 3 – Change to water regime

The indicators comprising WT's PC3 assess changes to water regime, as a pressure, by analysing land use in wetland catchments and by recording the presence or absence of river regulation upstream of wetlands (P16), the direct abstraction of water from wetlands (P19) and indirect changes to wetland hydrology through clearing native vegetation in the wetland's 1km buffer (P14).

Water regimes are regarded by many aquatic ecologists as the key driver regulating the ecological processes and diversity of floodplain wetland ecosystems, with alteration of flow regimes often claimed to be the most serious and ongoing threat to the sustainability of rivers and their associated floodplain wetlands (Allan and Johnson 1997; Bunn and Arthington 2002; Rolls et al. 2018; Adame et al. 2019).

Boulton et al. (2014) describe the main spatial and temporal features of water regime as timing, frequency, duration, extent, volume, depth, and variability. A floodplain's water regime is a multi-dimensional and complex system involving total flow, seasonality of flow, hydroperiod (wet/dry cycles), connectivity and distribution patterns, and the areal extent of inundation (Kingsford 2000; Thoms et al. 2005; Karim et al. 2014; Rolls et al. 2018). All aspects can vary naturally, resulting in a very dynamic ecosystem and high biodiversity through the rich availability of ecological niches (Kingsford 2000; Karim et al. 2014; Rolls et al. 2018). Non-floodplain wetlands, while arguably less hydrologically complex, are also influenced by seasonality, wet/dry cycles and frequency of connection to natural water sources (State of Queensland 2010, 2011a). A key factor in ecosystem function is the degree to which a wetland's waterbody is permanent or ephemeral. Ephemeral wetlands, filling and emptying on different time scales, predominate in mainland Australia (Lake and Bond 2007) and, in the GBRCA, are among the most vulnerable to being lost or degraded (GBRMPA 2012).

Rolls et al*.* (2018) note that water has three fundamental mechanistic roles in ecosystems. These are: acting as a vector for connecting and moving energy, material and organisms; disturbing ecosystems and contributing to geomorphic change; and acting as a habitat or resource for biota. As a resource for humans (post European settlement) vast amounts of water have been removed from ecosystems with major impacts on water regimes and biota that depend on natural conditions (e.g. Kingsford 2000; Thoms et al. 2005; Whalley et al. 2011).

People alter wetland water regimes, intentionally or inadvertently, to achieve a variety of purposes. Water is taken, whether directly from wetlands or from their catchments via impoundments (major and minor) for human and animal consumption, crop watering, industrial washdown processes, mining and other uses. People also use and modify wetland and floodplain channels, as part of agricultural distribution systems, alter natural flows for flood mitigation and interfere with flow regimes while generating hydroelectricity. Water regimes can also change as a by-product of other catchment activities, such as clearing forests and planting crops, both of which can affect the volume of runoff to wetlands (Cheng and Yu 2019).

As a result, volume, frequency and direction of surface and underground flows are changed (Thoms et al. 2005; Frazier and Page 2006; Steinfeld and Kingsford 2013; Tulbure and Broich 2019). In most Australian systems, people's use of water results in drier conditions (terrestrialisation); while with river regulation, flow patterns are also likely to be changed (Lake and Bond 2007). Wetlands that experienced extreme cycles of wet and dry through natural patterns of flood and drought, prior to human-induced changes to its water regime, will often be maintained at more stable water levels. In some cases, formerly ephemeral wetlands are used as storage ponds in irrigation systems, making them permanent, deep waterbodies (Perna et al. 2012b).

The ecological impacts on wetland condition of these water use-related actions are complex, often involving interactions with other classes of pressures (pest introductions, habitat modification and inputs) and interactions among the wetland environmental values impacted. In general, alterations to water regimes tend to reduce diversity of wetland plants, macroinvertebrates and aquatic vertebrates while favouring exotic and pest species (Catford et al. 2011).

Indicator P14 – Altered surface flow due to vegetation cleared

Indicator P14 contributes to PC3 and indirectly assesses the impact of clearing on individual wetlands by mapping the extent of remnant and regrowth vegetation in the wetland's 1 km buffer.

Vegetation clearing influences overland and subterranean water flow (Brookes et al. 2017) affecting wetland hydrology by altering runoff into the system. Herron et al. (2002) and Woodward et al. (2014) both note that trees reduce the amount of water available to wetlands, therefore deforestation increases the water yield to wetlands. According to Woodward et al. (2014), this *can convert ephemeral swamps to permanent lakes or even create new wetlands. This effect is globally significant, with 9 to 12 percent of wetlands affected, including 20 to 40 percent of Ramsar wetlands, but is widely unrecognized because human impact studies rarely test for it.*

In the mid-1960s, 45 percent of the Comet catchment in central Queensland, was cleared. A modelling study of the clearing showed an increase of up 40 percent of annual streamflow (Peña-Arancibia et al. 2012; Siriwardena et al. 2006). However studies on this catchment have had inconsistent findings with respect to the effects of clearing on stream hydrology. Investigations into these conflicts by Cheng and Yu (2019) showed that the effects of land clearing were unmistakeably detectable, with greater amounts of land clearing producing larger effects on flows. They concluded that previously inconsistent findings were likely due to differences in methods.

Stream flow increase after forest cover decrease has also been reported in Java Indonesia and central Brazil (Costa et al. 2003, Peña-Arancibia et al. 2012). Peña-Arancibia et al. (2012) argued that 'small' catchments (< 10,000 km^2) do not always show an increase in streamflow after vegetation clearing, although Citarum River in west Java, Indonesia (4133 km^2 catchment) had a 50 percent decrease in forested cover and an 11 percent increase in stream flow after vegetation was cleared in its catchment (Peña-Arancibia et al. 2012). They also noted that increased runoff in some small catchments may be attributed to development of hard surfaces such as roads and buildings (Peña-Arancibia et al. 2012) in addition to forest clearing. Woodward et al. (2014) found that forest clearing led to a significant increase in catchment water yield (18 to 28 percent) and speculated that this may be an *underestimation of the actual magnitude of the hydrological legacy of deforestation*. They further suggested that the hydrological changes resulting from vegetation clearing will depend on catchment-to-wetland area ratios (the higher the ratio, the greater the impacts) and that impacts will be greater in shallower and more ephemeral wetlands (Woodward et al. 2014).

Indicator P16 – Change in landscape hydrological integrity

Indicator P16 contributes to PC3 and assesses change in landscape hydrological integrity within a 1 km radius of the wetland, and has two components, (a) changes to floodplain hydrology through specified water management and river regulation and (b) pressure associated with land use surrounding the wetland.

Since European settlement in the 1850s, substantial changes in catchments, through the construction of weirs, floodplain levee banks and large dams (Arthington and Pusey 2003), have led to alterations in the quality and quantity of water reaching the GBR (Waterhouse et al. 2016). The hydrology of major rivers has changed across temporal scales due to flow regulation, which *is widely acknowledged to be a major cause of deteriorating conditions in many Australian river and floodplain ecosystem*s (Arthington and Pusey 2003). A river's flow regime sets into motion a host of ecological processes, with floodplain wetlands being fundamental to ecological functioning (Kingsford 2000; Dyer 2002). Specific features of the flow regime (flow permanence and regularity; flow variability and absence; wet–dry seasonality) have been proposed as the *key hydrological drivers in maintaining and explaining the ecological function and biodiversity of large rivers in Australia's north, including their associated wetlands and floodplains* (Leigh and Sheldon 2008).

River regulation and catchment scale water extraction can alter the water and flow regime of wetlands and streams by altering their wet/dry cycles. Some wetlands that historically dried are now permanently wet. Others are becoming drier (terrestrialisation) and in some instances they have lost their natural seasonal patterns (Arthington and Pusey 2003; Catford et al. 2011; Lunt et al. 2012). According to Kingsford (2000), Watkins et al. (2011) and Catford et al. (2014), increasingly managed river systems have directed riverine ecosystems to become more terrestrial worldwide. River regulation has led to extensive deterioration in the extent and health of wetlands (Fu et al. 2015) and hydrological modification, through catchment scale water extraction, has altered species composition and biodiversity and given exotic species greater opportunities to colonise (Bunn and Arthington 2002; Catford et al. 2011, 2014).

Land use is also linked with changes in wetland water regimes, with consequences for wetland environmental values. The links between alterations to hydrology and land use surrounding wetlands are complex and involve interactions between change to water regime and other pressures (Bunn and Arthington 2002; Lunt et al. 2012; Hamer and Parris 2013; Kath et al. 2014). Land use-associated direct pressures include surface and groundwater abstraction for stock watering and mining, and the construction of dams, levees and channels for a variety of purposes including irrigation of pastures and crops, stock watering and aquaculture (Kingsford 2000; Frazier and Page 2006; Steinfeld and Kingsford 2013; Waterhouse et al. 2016). Surface flows can be altered to varying degrees by the construction of roads and railways, preparation of the ground for intensive cropping or plantation forestry and alterations to topography for various land uses including housing, agriculture, grazing and mining (Kingsford and Thomas 1995; Kingsford 2000; Raiter et al. 2018; Tulbure and Broich 2019). Urban development brings impervious surfaces, channelling and rain harvesting as well as surface water abstraction and to a lesser extent, the abstraction of groundwater, which can all alter surface flows (DSITIA 2015).

Indicator P19 – Abstraction (water taken out for use) or consumption by livestock and feral animals

Wetland ecology is dependent on the water regime, as it influences biotic patterns in space and time. Indicator P19 contributes to PC3 and assesses abstraction (i.e. the taking of water for use) and/or consumption of water by livestock or feral animals, which can affect the extent, depth and duration of saturated or ponded conditions within a wetland (i.e. features of the water regime).

Human-induced alteration to patterns of drying and inundation is widespread throughout Australia (Brock and Crossle 2002; Dyer 2002; Jenkins and Boulton 2007; Karim et al. 2014). Some wetlands have become permanent water storages, altering their species diversity and composition (Dyer 2002; Arthington and Pusey 2003). In other wetlands, the taking of water for use (abstraction) has resulted in systems becoming drier (Catford et al. 2011), affecting ecological status, impacting biodiversity and altering wetland function (Acreman and Miller 2006). Water extraction can also alter the time between inflows from connected wetlands, often increasing this time, with consequent impacts on fish, waterbirds, microbial activity and vegetation (Jenkins and Boulton 2007). Reduced flow due to abstraction may also alter the movement of sediment in floodplain systems, increasing sedimentation in the wetland (Reid et al. 2018).

The variable hydrological cycles of wetting and drying of wetlands, which differentiates them from exclusively aquatic or terrestrial ecosystems, means that even putative trivial changes, such as minor water abstraction, can result in significant changes to wetland processes and functional relationships (Acreman and Miller 2006).

Pressure class (PC) 4 – Input pressures

The PC4 subindex assesses the pressure of land use associated with pollutant inputs, including nutrient, sediment, septic system-derived, pesticide and stormwater inputs (Chandrasena and Sim, 1999; Rippy et al. 2017), within the 200 m or 1 km radius of a wetland, depending on the type of input considered. This aligns with the landscape scale assessment of land use hazard to wetlands in the GBR catchments (DSITIA 2015), which attributed nine direct and indirect pressures to PC4. These were nutrients, sediment, pesticides, chemicals and metals, organic matter, salt, acids, hot and cold water, and litter and rubbish.

In the GBRCA, 87 percent of the land is modified for cattle grazing, intensive cultivation, dairy, sugarcane, horticulture and mining, causing inputs to wetlands of sediments, nutrients, pesticides and other pollutants via surface runoff and groundwater drainage (Adame et al. 2019). These pollutants have direct impacts on wetlands and on downstream ecosystems, including the GBR lagoon (Lewis et al. 2009; Pearson et al. 2013; Arthington et al. 2015; Waterhouse et al. 2017).

Pollutants are introduced into wetlands either as runoff of substances applied in the course of land management, for example, pesticides and fertilisers (a source of nutrients), or they are mobilised and introduced into ecosystems through land use related changes to natural flows, hydroperiod and other dimensions of water regimes. Changes to vegetation surrounding wetlands can also facilitate the input of pollutants including sediment, nutrients, organic matter, and litter and rubbish by affecting the filtering functions of the riparian zone (Karr and Schlosser 1978; Naiman and Décamps 1997; Waltham et al. 2019). Consequently, the pressure class PC4 *Inputs* is moderately correlated with both PC3 *Change to water regime* and PC2 *Habitat modification*. These findings from the analysis of baseline WT data collected in 63 freshwater floodplain wetlands in the GBRCA are supported in wetland literature with numerous examples of interactions among the three classes of anthropogenic pressures (e.g. Faulkner 2004; Lee et al. 2006; Rassam and Pagendam 2009; Kath et al. 2014; Adame et al. 2019).

The magnitude of pollutant impacts on wetlands is determined by pollutant loads, duration of exposure, wetland area and climate (Adame et al. 2019). Pesticides, nutrients, sediments and other sources of direct input pressures primarily affect the wetland's biotic integrity (Bolger and Stevens 1999; Chandrasena and Sim 1999; Liess and Schulz 1999; Reid et al. 2007; Gell and Reid 2014; Belmer et al. 2018; Adame et al. 2019) while sediment is also implicated in changes to the hydrology and aquatic connectivity of wetlands through the ecological processes of sedimentation, erosion, deposition and mobilisation (Gell et al. 2009; GBRMPA 2012).

Indicator P3 – Land use associated pesticide residue inputs

Indicator P3 contributes to PC4 and assesses pressure on wetlands in terms of land use within wetlands and their 1 km buffer zone that is associated with pesticide residue inputs. Pesticides include insecticides, herbicides and other pest control chemicals. Pesticides vary considerably in their chemical properties, which in turn affects their persistence in the environment (Thorburn et al. 2013).

The use of pesticides is both widespread (i.e. not restricted to agricultural settings) and varies considerably in regard to the types, concentrations, volumes, applications, and spatial and temporal distribution of pesticides used (Adame et al. 2019). Land use in the GBRCA associated with high pesticide use includes intensive cropping and horticulture and intensive animal production (DSITIA 2015). In sugarcane and grain cropping regions of the GBRCA, herbicide applications are common (Thorburn et al. 2013). Kennedy et al. (2012) note that photosystem II (PSII) herbicides are the most commonly used and ecologically significant pesticides detected in the GBR marine environment.

Agricultural plant canopies, crop residues and soils treated with pesticides are subject to wash-off and leaching by rainfall and irrigation (Thorburn et al. 2013). However, pesticides are not restricted to use in agriculture and have become more common in urban and other high use areas, particularly in structural pest control, landscape maintenance and residential home and garden applications (Jeppe et al. 2017). Preda et al. (2006) note that overland flow events in timber plantations can mobilise soil additives including nutrients and herbicides, which can then enter waterways, including wetlands.

Vandergragt et al. (2020) examined the presence and concentrations of pesticides in 22 lowland floodplain wetlands surrounded mainly by high to moderate intensity land uses, including sugarcane growing, grazing and plantation forestry. They found 59 pesticides and pesticide breakdown products in two early wet season sampling periods, with an average of 21 pesticides detected per wetland. Statistically significant relationships were found between the number of pesticides detected and both rainfall in the two months prior to sampling and the percentage of sugarcane as a land cover within 1 km of the wetland.

Devlin et al. (2015) comment that *studies of pesticides in GBR freshwater wetlands are notable for their absence in the published literature*. They speculate about the behaviour and impact of pesticides in freshwater wetlands, given the highly variable hydrology of those wetlands. They single out coastal freshwater wetlands as receiving the highest impact from pesticides of all aquatic ecosystems in the GBRCA. Adame et al. (2019) characterise freshwater wetlands in the GBRCA as *threatened by extensive pulses of floodwater carrying high pollutant loads during the wet season, and by chronic exposure to poorly flushed water during the dry season*. They attribute a range of impacts to PSll inhibitors, the most regularly reported herbicides in the GBRCA. Impacts include reduced growth in benthic macroalgae, changes to diatom community composition, physiological disturbances in amphibians and fish, and changes in the dynamics of fish populations. Liess and Schulz (1999) also demonstrated that agricultural insecticides alter the dynamics of macroinvertebrate communities in freshwater streams.

Indicator P4 – Land use associated with nutrient inputs

Indicator P4 contributes to PC4 and is similar to P3, except that it assesses land use associated with nutrient rather than pesticide inputs. Excess nutrients commonly enter streams and wetlands via runoff of agricultural chemicals, including fertilisers, which is a major concern due to their typically negative impacts on aquatic ecosystems (Connolly et al. 2015). These include rapid growth in some plant species followed by rapid die off; the mass of decomposing plants in the water causes deoxygenation which can result in morbidity or mortality of other life forms. Also, one plant species may propagate in unusually high numbers, stifling the growth of other plants by blocking access to sunlight and restricting movement of aquatic animals through the water. An excess of nutrients can also help invasive plants establish by providing them with nutritional resources they need but which native flora requires at lower levels (Adame et al. 2019).

Fine sediments and attached nitrogen, mainly attributed to grazing lands, and dissolved inorganic nitrogen, mainly associated with cropping, can enter the GBR lagoon from catchments draining agricultural lands (Thorburn et al. 2013). Tsatsaros et al. (2013) identified that nutrients from erosion and fertiliser use in the Wet Topics region of Queensland are a major issue and that these, along with associated environmental issues such as turbidity and reduced dissolved oxygen, mainly arise from agriculture, more so than urban development. The study notes that concentrations of dissolved inorganic nutrients in floodwaters from less developed catchments or rangeland grazing areas are much lower than concentrations from waters discharging through cropping areas and urban areas in the lower, coastal catchments.

The average intensity of nitrogen fertiliser application varies substantially across the various catchments of the GBR, predominantly depending on the type of agriculture present. Intensive agriculture and sugarcane use up to twice as much nitrogen fertiliser as other forms of agriculture (Thorburn et al. 2013). Waterhouse et al. (2012) found that in the Wet Tropics region, the sources of dissolved inorganic nitrogen (DIN) are 75 percent from sugarcane, 5 percent from bananas, 12 percent grazing and forests, and 8 percent from other crops, dairy and urban areas.

Impacts of nutrient contamination on wetland environmental values can range from changes in diatom assemblages, detectable on paleolimnological timescales (Gell and Reid 2014), to phytoplankton blooms leading to acute hypoxia and fish kills (Pearson et al. 2013; Davis et al. 2017). The main ecological responses to nutrients in freshwater systems are increased plant production, weed infestation, increased hypoxia and plankton blooms (Pearson et al. 2013; Davis et al. 2017). Weed infestations typically make minor contributions to aquatic food webs despite their high biomass, and starve the productive bed of light and prevent reoxygenation of the water column (Verhoeven et al. 2006). Flow on effects to fauna include decreased macroinvertebrate and fish diversity (Arthington et al. 2015; Adame et al. 2019).

As nutrient loads persist and increase, wetland systems can tip from stable states with low turbidity and abundant submerged macrophytes to eutrophic, turbid states with prolonged phytoplankton dominance. However, Pearson et al. (2010), in a Wet Tropics case study, highlight the complexity of predicting the impacts of land use generated contaminants on freshwater systems and biota. For example, hypoxia from nutrient contamination may not occur if the water is turbid enough from sediment inputs to limit phytoplankton growth.

Indicator P5 – Number of septic systems within 200 m of wetland per ha of mapped wetland

Indicator P5 contributes to PC4 and estimates the number of septic systems within 200 m of wetlands to assess the influence of pressures due to on-site sewage/wastewater treatment sites, and their density, on wetland condition. Beal et al. (2005) indicate that as septic tank density increases, so does the potential for cumulative water contamination, and that there is a clear hydraulic link between septic systems and nutrient contamination of surface and groundwater. This suggests that impacts on wetland condition due to septic tanks (at the 200 m buffer scale) may be similar to impacts due to nutrient inputs (at the 1 km buffer scale).

Approximately 12 percent of the Australian population use on-site sewage/waste-water treatment (septic) systems (Geary and Gardner 1998). The potential for nitrate contamination of groundwater from unsewered areas (that rely on septic systems) is higher than from sewered areas because nitrogen loading rates in unsewered areas is higher (Bolger and Stevens 1999). Geary and Whitehead (2013) concluded that there is a definite relationship between the quality of shallow groundwater aquifers and onsite septic systems, particularly when such systems are at higher densities. Nitrate in groundwater can pose a risk to the environment when discharging into receiving waters (such as wetlands) as it alters the nitrogen to phosphorus ratio, leading to a greater risk of eutrophication and algal blooms (Bolger and Stevens 1999).

Indicator P12 – Number of stormwater or other point inflows per hectare of wetland

Indicator P12 contributes to PC4 and measures all types of engineered structures channelling water into wetlands, including drains, ditches and culverts. Stormwater systems are constructed to manage the increase in stormwater flows resulting from an increase in impervious surfaces (i.e. areas where rainwater cannot infiltrate into the ground) in catchments. Impervious surfaces increase runoff and flows to receiving wetlands and are generally linked with urbanisation (Faulkner 2004). This stormwater-induced increase in flow to wetlands is one way urbanisation modifies the water and sedimentation regimes of wetlands as well as the dynamics of nutrients and other pollutants. Structural and functional ecological changes follow (Lee et al. 2006). Decreased sinuosity associated with stormwater infrastructure also results in increased velocity of stream discharge to receiving wetlands (Lee et al. 2006).

Faulkner (2004) notes that when a wetland’s water regime is significantly altered by urban land use, the alteration is usually permanent, with a tendency to move still further away from the natural state. Faulkner (2004) concluded that increased nutrient and contaminant loads due to wetland hydrology changes lead to changes in plant species composition, ultimately impacting species diversity and abundance in avian, amphibian and macroinvertebrate populations. Belmer et al. (2018) indicated that, within upland swamps of the Blue Mountains Region of New South Wales, there was a notable reduction in species diversity of aquatic invertebrates and a higher pH in swamps located within urbanised catchments compared to those in non-urban areas. A study of wetlands in Botany, New South Wales, found that over a period of decades, native vegetation was degraded and replaced by exotic species due to changes in nutrient loads and other pollutants from urban and industrial stormwater (Chandrasena and Sim 1999).

Indicator P10 – Sediment supply (modelled, GBR)

Indicator P10 contributes to PC4 and assesses sediment supply into wetlands, based on modelled fine sediment rates for land use groupings (QLUMP).

Sedimentation and water regimes are key physical drivers in wetlands, and both are frequently modified by human activities, often through the process of urbanisation (Lee et al. 2006). Increased sediment loads to wetlands have several impacts including a reduction of available light, increased nutrient and contaminant loads and alterations to the soil substrate. As well, sediments that flow into wetlands transform sites from being topographically heterogeneous into flat plains that support reduced biodiversity and encourage weeds (Zedler and Kercher 2004). Grazing and irrigated cropping are known to be responsible for a broad range of stressors on aquatic ecosystems including increased fluxes of sediment and nutrients from terrestrial to aquatic ecosystems as well as hydrological changes (Reid et al. 2018). Sediment loads entering wetlands have substantial impacts on the growth and abundance of vegetation within wetlands. Reid et al. (2007) note that an increase in nutrient levels and change in transparency and substrate would likely have resulted in aquatic plant loss on a large scale in billabongs of the Murray-Darling River system. Burton and Johnston (2010) note that the loading of chemical contaminants in aquatic systems is generally from diffuse sources (i.e. eroded sediments).

Gell et al. (2009) studied 20 wetlands in the Murray-Darling Basin and concluded that sedimentation rates increased in all wetlands following European settlement, resulting in increased sedimentation rates of up to 80 times the mean rate. This led to complete infilling of some wetlands through the Holocene, with some transforming to a terrestrial state. Within the same system it was found that, as a consequence of human activities, lagoons and billabongs have been impacted by increased nutrient and salinity loads and increased sedimentation rates, resulting in higher turbidity and loss of macrophytes (Gell and Reid 2014). A paleolimnological study of two wetlands of the Burdekin River, Labatt Lagoon and Swann’s Lagoon, found both had historically experienced significant increases in sedimentation as a result of land clearance (Tibby et al. 2019). Similarly, Hanson et al. (2022) found increased sedimentation and shallowing of two wetlands (Lake Mary North and Tualka) associated with land use change in the Fitzroy Basin region.

Indicator links to wetland condition: state index

Wetland Environmental Value (WEV) 1 – Biotic integrity

WEV1 assesses the biological health and diversity of wetland ecosystems, in other words, their biotic integrity. It comprises two indicators (S1 and S3) that assess the vegetation composition and structure and exotic plant species cover of wetlands and their 200 m buffer zones.

The concept of naturalness is well established in ecological assessments, where it is broadly defined as a lack of human-induced disturbance. It includes ecological integrity, or the capacity of an ecosystem to maintain its natural properties and functions and remain resilient to natural forms of disturbance (DEWHA 2008)**.** *Biotic* integrity is associated with how natural or 'pristine' a wetland and its function are relative to the potential or original state before intensification of land use.

While water regimes are the key drivers of floodplain wetland ecosystems, vegetation is key to a wetland's overall biotic integrity and its floristic identity and habitats. As such, wetlands in Queensland are classified and mapped by key attributes, including vegetation, that define wetland types e.g. coastal and subcoastal floodplain grass, sedge and herb swamps or wet heath swamps or floodplain tree swamps – melaleuca and eucalypt, etc. (DES 2019a). Palustrine (swamp) wetlands in particular are characterised by vegetation, namely the presence of tree, shrub, and emergent plant communities.

Indicator S1 – Floristic composition and vegetation (community) structure

A key indicator of change in wetland biotic integrity is the state of wetland, riparian and buffer zone plant communities (Boyd 2001; DERM 2011; Shackleton et al. 2019). As such, Indicator S1, which contributes to WEV1, assesses vegetation composition and structure of wetlands and their 200 m buffer zones, and the degree to which the vegetation is consistent with mapped preclearing and remnant RE descriptions.

Plant communities are often used as biological indicators of wetland condition because they are among the best predictors of multiple aspects of overall wetland condition (e.g. Rooney and Bayley 2012, Stapanian et al. 2015) and biotic integrity, including for example: amphibian biotic integrity (Stapanian et al. 2015), fish biotic integrity (Cooper et al. 2018), and waterfowl and invertebrate assemblages (Rooney et al. 2012). In Fitzroy River backwaters (Queensland, Australia) a significant reduction in wetland native plant diversity is implicated in reducing macroinvertebrate abundance and influencing fish abundance and composition, with potential to impact on waterbird habitat values of wetlands (Houston and Duivenvoorden 2002). In prairie wet meadows (Canada), vegetation and songbirds are strong surrogates for one another suggesting wet meadow vegetation condition can be used to predict the health of wetland-dependent songbirds (Wilson and Bayley 2012). Similarly the perennial grass (water couch) meadows of eastern Australia are significant habitat for migratory waterbirds (Price et al. 2011) and thus integral to the biotic integrity of floodplain wetlands.

Other vegetation important to wetland biotic integrity is the vegetation occurring in the terrestrial buffer zones surrounding wetlands. These are core habitats for many semi-aquatic and wetland dependent species (Boyd 2001; Semlitsch and Bodie 2003; DERM 2011). They also provide wildlife habitat connectivity and influence wetland ecosystem processes (Boyd 2001; Semlitsch and Bodie 2003). The width of buffers and their vegetation composition are key features that support and enhance the functions of healthy wetlands (Ma 2016).

Biological indicators have been widely tested and used to indicate wetland disturbance and biotic integrity. However they often require detailed inventory, can be complex and are time consuming (e.g. Lopez and Fennessy 2002; Mack 2007). Nevertheless, plant-based assessments have been shown to correspond linearly to other reference measures that quantify individual and aggregate pressures (e.g. Kutcher and Forrester 2018). Rooney and Bayley (2012) show that many of the common wetland bioindicator assemblages respond to the same environmental gradients. They argue a single community (e.g. a plant community, or a fish community) can serve as a surrogate for other communities (e.g. bird and reptile communities) and should be adequate to estimate the underlying environmental conditions and the degree of disturbance of the ecological community as a whole (Rooney and Bayley, 2012). Gara and Stapanian (2015) show that two variables often used in plant-based bioassessments – species presence and cover – assess critical ecosystem elements well enough to determine wetland condition and overall biotic integrity.

Indicator S3 – Exotic plant cover

Indicator S3 contributes to WEV1 and assesses the presence and percentage cover of non-native (i.e. exotic) plant species that may be present in a wetland and its 200 m buffer zone.

The native vs exotic origin of plant species is one of the most strongly related plant traits to land use driven habitat disturbance and associated changes to the physical and chemical properties of wetlands (Roy et al. 2019). This is because habitat disturbance facilitates the spread of exotic plants (MacDougall and Turkington 2005; Mayor et al. 2012). Land use associated habitat disturbance facilitating the establishment of exotic plant species can include introductions for agricultural purposes (Housten and Duivenvoorden 2002; Wurm 2007), alteration of hydrology (Catford et al. 2011), overgrazing grazing by cattle (Wurm 2007; Lund et al. 2012), soil disturbance (Price et al. 2011), dumping and garden escaped plants (Stephens and Dowling 2002).

As outlined above in the Pressure Class sections of this literature review, exotic plant species can dominate and outcompete native plant species (Rea and Storrs 1999; Hastwell et al. 2008) altering the habitat conditions for native plants and other biota (Douglas and O'Connor 2003). Invasive exotic plants in particular can alter the ecological functions of whole ecosystems including community structure (MacDougall and Turkington 2005), productivity and trophic structures, nutrient cycling and hydrology (Vitousek 1990). Exotic species have also been shown to be ecological traps (Stinson and Pejchar 2018), to negatively influence the reproductive success of songbirds that use introduced plant species within wetland habitats (Stinson and Pejchar 2018), and to effect site fidelity and age ratios of migratory birds thus altering song-learning conditions (Ortega et al. 2014). Other studies show altered eco-exergy (biomass plus genetic information) attributes of macroinvertebrate communities in mangrove wetlands (Chen et al. 2018), and reductions in wetland native plant species diversity and macroinvertebrate abundance, leading to changes in fish abundance and composition, and potentially impacting waterbird habitat values of wetlands (Houston and Duivenvoorden 2002).

While in some circumstances, there may be a positive or neutral association between non-native plants and native species (Sax 2002; Martin-Fores et al. 2017; Ward-Fear et al. 2017; Guerin et al. 2018; Stinson and Pejchar 2018), evaluations of floristic quality indices overwhelmingly conclude that non-native plant indicators should be included in wetland condition indices. There is strong agreement that the presence or absence of non-native plants provides a clear indication of wetland ecosystem condition, the biotic integrity of a wetland and its function relative to the potential or original state before intensification of land use was imposed (e.g. Ervin et al. 2006; Miller and Wardrop 2006; Kutcher and Forrester 2018; Roy et al. 2019).

Wetland Environmental Value (WEV) 2 – Local physical integrity

WEV2 assesses a wetland's local physical integrity and comprises two individual indicators (S7 and S8) that assess geomorphic and soil disturbance to wetlands and their buffer zones by humans, livestock and/or feral animals.

Wetlands are dynamic ecosystems that develop and change in response to changing external (e.g. geological, climatic) and ecological conditions (e.g. vegetation succession, animal activities) or internal geomorphological shifts that influence erosional and depositional patterns. These patterns of change occur naturally over different temporal and spatial scales (Tooth 2018). Extreme weather events and some human activities have the potential to drive rapid changes.

The distinctiveness and diversity of geomorphic or micro-topographic features are among a range of wetland attributes susceptible to disturbance (AETG 2012). These abiotic features – the landform and distribution and retention patterns of sediment within a wetland – are integral components influencing wetland hydrologic and biotic processes (AETG 2012; Kim and Kupfer 2016). Wetland soils store nutrients important for primary production and enable nutrient cycling and are the physical layer that supports aquatic plants (macrophytes and algae) and provides habitat for benthic invertebrates and microorganisms.

Indicator S7 – Direct disturbance by humans, livestock of feral pests impacting soil

Indicator S7 contributes to WEV2 and assesses direct disturbance to wetlands and their 200 m buffer zones by humans, livestock or feral animals that exposes the soil surface making it more susceptible to erosion or sediment mobilisation.

Disturbed and exposed wetland and buffer zone soils indicate erosion and sedimentation may be occurring. Sedimentation resulting in filling of wetlands is among the most significant runoff-associated threats to wetlands (Gleason and Euliss 1998; Skagen et al. 2008). Although the movement or erosion of soil by rain and flooding is a natural part of the erosional and depositional processes affecting wetlands, these processes can be intensified by anthropogenic disturbance and the resultant exposure of local soils.

The majority of GBRCA wetlands are embedded within an agricultural landscape (DES 2019b) where land uses have significantly changed surface runoff patterns at both catchment and local scales, causing surface runoff of sediment and other pollutants to flush into wetlands.

Vehicle and livestock tracks and cultivation can expose and harden wetland and wetland buffer zone soils (Luo et al. 1997; Gleason and Euliss 1998; DEC 2012; Bowen and Johnson 2017; Leon et al. 2017). Shallow channels can form that cut through critical water-flow regulating points resulting in faster, concentrated water flow. This faster flow, in turn, increases soil erosion and can lead to accelerated sedimentation and shallowing of wetlands (DEC 2012; Bowen and Johnson 2019). Similarly, livestock and feral animals directly accessing wetlands and drainage lines disturb and expose soils leaving banks susceptible to slumping and subsidence (Cassanova 2007; Queensland Government 2011).

In summary, direct disturbance by humans, livestock or feral pests that impacts soil, such as vehicle and livestock tracks, cultivation and other human induced exposure of soils in the wetland and wetland buffer zone, indicate changes in hydrology and consequently, potentially altered erosional and deposition processes that change a wetland's local physical integrity and overall wetland condition.

Indicator S8 – Soil surface deformation from livestock or feral pests in the mapped wetland

Indicator S8 contributes to WEV2 and assesses deformation of a wetland's soil surface (i.e. substrate) from pugging, tramping, digging and/or wallowing by livestock or feral animals.

The condition of a wetland's substrate, topography and physical form is widely recognised as an important component of overall wetland biophysical integrity. As such, indicators of substrate disturbance and compaction, including damage by livestock and feral pests, are commonly included in wetland assessment methods (e.g. Clarkson et al. 2003; Harding 2005; Papas 2005; Fennessy et al. 2004, 2007; Herlihy et al. 2019). This is because the saturated or near-saturated soils of wetlands have low mechanical strength, making them susceptible to physical damage (Eyles 1977; Evans 1998; Askey-Doran and Pettit 1999). Pugging, trampling, rooting (digging) and wallowing are forms of direct physical damage to a wetland's substrate and micro-topography and are clear indicators that a wetland's physical state is compromised (e.g. Davis et al. 2017), being indicative of altered or destroyed substrate cracks, changes in the size and number of interstitial (air) spaces in the soil, increased soil density, decreased soil porosity and reduced infiltration of water and oxygen (Holmes et al. 2009; Crush and Thom 2011; Price et al. 2011; Morris and Reich 2013; Marshall et al. 2020). Pugging and trampling are caused by livestock and hard hooved feral animals (e.g. cattle, horses, goats, pigs) continually moving across damp or wet soils and causing soil compaction (Holmes et al. 2009; Crush and Thom 2011).

Substrate disturbance can also be a predictor of other disturbances to overall wetland biophysical integrity, including for example, compromised aquatic and biotic diversity (Cumberlidge et al. 2009; Whyte and Anderson 2017; Marshall et al. 2020), disruption of seed and egg banks, inhibited terrestrial and aquatic plant re-establishment, destruction of benthic algae fringes (Pettit et al. 2012; Morris and Reich 2013) and changes to water quality (Askey-Doran and Pettit 1999; Papas 2005; Cassanova 2007; Pettit et al. 2012). Other potential consequences of substrate disturbance are reduced water-storage capacity, altered water temperatures with increased evaporation (Kihwele et al. 2012; Davis et al. 2017) and increased potential for erosion (Cassanova 2007; Peters et al. 2015).

Wetland Environmental Value (WEV) 3 – Local hydrology

WEV3 indicators are designed to indirectly assess the state of wetland (i.e. local) hydrology at different spatial extents (wetland and 200 m buffer, S9; 1 km buffer, S15; and wetland and 1 km buffer, S16).

As summarised in the Pressure Class sections of this literature review, hydrology is a key driver of wetland ecosystem processes and hence, wetland condition (Boulton et al. 2014; Mitsch and Gosselink 2015). Departure from a wetland's natural hydrological state, at all spatial and temporal scales, can therefore perturb wetland environmental values. Changes can range from subtle shifts in sensitive aquatic biota (Bunn and Arthington 2002) to complete loss of wetlands, along with their associated values (Kingsford 2000).

A wetland's hydrology is multi-dimensional and complex, involving total flow, seasonality of flow, hydroperiod (wet/dry cycles), connectivity and distribution patterns, and the areal extent of inundation (Kingsford 2000; Thoms et al. 2005; Karim et al. 2014; Rolls et al. 2018). Hydrology also interacts with local geomorphology to determine details such as wetland depth profiles and the velocity of flow through different parts of the wetland (e.g. State of Queensland 2011b; 2011c). Each of these hydrological components and processes is influenced at multiple scales by human land management, including diverting and storing water for upstream use but also alterations to natural surface flows across floodplains and within individual wetlands and their natural inlets and outlets (Leigh et al. 2010).

Indicator S9 – Drainage modification and artificial structures altering natural surface flows

Indicator S9 contributes to WEV3 and assesses wetlands and their 200 m buffer zones based on the number of drainage modifications and artificial structures present that, locally, indicate an altered state of wetland hydrological processes. Drainage modifications and structures include excavation or infilling, earth banks constructed within the wetland, earth works and artificial structures causing pinch points in surface flow, ditches, drains, gutters and canals that divert water to or away from wetlands, and earthworks or structures that modify or impede natural inlets or outlets of the wetland.

Drainage modifications and artificial structures that alter natural surface flows have potential impacts on each of the components and processes of a wetland's local hydrology. Total flow, seasonality of flow, hydroperiod, connectivity and wetland extent can all be affected by earthworks constructed in or around a wetland for various land use purposes. As well, local variables such as water depth and movement within a wetland, can be altered. For example, even minor infilling or excavation of parts of a wetland can alter the depth profile or velocity of flow, affecting whether, or how, biota occupy the wetland (Bunn and Arthington 2002; Acreman and Miller 2006; Rolls et al. 2018).

Bunn and Arthington (2002) emphasise that aquatic species evolved life history strategies primarily in response to natural flow regimes and Rolls et al. (2018) suggest that this happens at multiple spatial scales. For example, decline in local species richness is linked with the loss of specialised habitats that are necessary for specialised taxa to persist (Rolls et al. 2018). Consequently, even low-level disturbances to surface flows into and through wetlands can cause change in the range and abundance of biota found in that wetland.

Halford and Fensham (2014) studied the relationships among vegetation and local environmental conditions in ephemeral subtropical wetlands in central Queensland, finding that water depth had the greatest impact on vegetation patterns, with bank slope having a secondary influence. The type and extent of vegetation will, in turn, influence the local faunal community, so minor excavation or infilling, while directly affecting the wetland's local physical integrity, will also have impacts on the biotic integrity of the wetland.

Within wetlands, barriers, for example earth banks constructed to carry roads or to isolate parts of the wetland, and pinch points, where water is constrained to flow through narrow openings rather than taking its natural course across the wetland, will alter water velocity, depth, slope, extent and other aspects of local hydrology linked with geomorphology and have ecological consequences (e.g. Acreman and Miller 2006; Steinfeld and Kingsford 2013; Waterhouse et al. 2016; Raiter et al. 2018).

Drains, ditches, canals and earth banks are often installed to direct water into or away from a wetland for various reasons. This will alter a wetland's extent and the duration of wet and dry phases, especially if lateral connections with floodplains and river channels are changed. The ecological health and biodiversity of many wetlands have declined as a result of such modifications to natural flows (Kingsford 2000, Rogers et al. 2012). In surface water ecosystems, Rolls et al. (2018) reviewed the complex links between biodiversity and hydrological regimes to better understand the effects of spatial scale. At the local level of biodiversity within a wetland, water permanence in wetlands was found to promote alpha diversity (local species richness) of diatoms, fish and macroinvertebrates, and fish species richness also increased with increasing persistence and depth of sites. In contrast, macrophyte species richness was greatest at intermediate or low inundation frequency or duration.

Earthworks in and around a wetland that change the hydroperiod, perturb natural seasonal effects and alter the depth and extent of wetlands all interfere with a wetland's lateral and longitudinal connectivity. A wetland can also be directly cut off from surrounding aquatic features by modifications that impede flow through its inlets and outlets. Sheaves (2009) in particular, but also Thoms et al. (2005), Leigh and Sheldon (2009), State of Queensland (2011c, 2011d), Rolls et al. (2018) and many others stress the importance of hydrological connectivity in maintaining a wetland's biotic integrity. Between and within wetlands and across floodplains, and as described in the section on pressures, connectivity plays a crucial role in ecological processes such as recruitment and regulating populations of aquatic organisms (Attum et al. 2008; State of Queensland 2011e; Adame et al. 2019), dispersal and migration of eggs, larvae and seeds of aquatic and riparian biota (Jenkins and Boulton 2003; State of Queensland 2011e), nutrient cycling and availability (Thoms et al. 2005; Meixner et al. 2007) and the downstream and lateral transport of nutrients (Thoms et al. 2005) among many other processes.

The presence of drainage modifications and human structures offers a surrogate measure of a wetland's hydrological state at the local scale as well as an indicator of the wetland's overall state. With hydrology a major driver of wetland ecological processes, alterations to flow brought about by land use related modifications will also disturb other wetland environmental values, such as the type, richness and distribution of wetland dependent and aquatic flora and fauna (WEV1, biotic integrity), the movement of sediment within and through the wetland (WEV2, local physical integrity) and a wetland's ecological and hydrological connection with surrounding ecosystems (WEV4, connectivity). Flows are a major determinant of physical habitat, which in turn is a major determinant of biotic composition (Bunn and Arthington 2002, Catford et al. 2011, Adame et al. 2019). Rogers et al. (2012) found that the requirements of individual wetland species were highly specific to hydrological variables. It follows that *any* change to wetland hydrology resulting from drainage modification or artificial structures introduced to control the direction and volume of wetland surface flows will affect a wetland's biotic integrity.

Indicator S15 – Modified and artificial wetlands

As a contributor to WEV3, Indicator S15 assesses modified and artificial wetlands and wetland features within the 1 km buffer of wetlands that could affect wetland hydrology.

Modified and artificial wetlands include bunded and dammed natural wetlands, excavated wetlands, constructed water storages such as farm dams, weirs and reservoirs, and channels or drains. Their presence in the landscape surrounding a natural wetland indicates that the hydrological state of that wetland is partly or completely changed.

Weirs, off-stream water storages and upstream flow diversions disrupt movement of water across floodplains, reducing the frequency and volume of flows to floodplain wetlands (Kingsford 2000). These reductions in hydrological connectivity can lead to longer dry phases in floodplain wetlands (Kingsford 2000; Steinfeld and Kingsford 2013). When individual wetlands are converted into off-stream water storages through damming, water levels typically become higher and more stable, with loss of natural wet-dry cycles and floodplain hydrological connectivity (Kingsford 2000). Farm dams and natural wetlands that have been altered to capture and store water for longer periods may not individually have much influence on overall system hydrology but the greater the number of these storages and the volume of the water stored, the greater the influence on the hydrologic state of connected wetland systems (Nathan and Lowe 2012).

Indicator S16 – Altered surface flow due to linear transport infrastructure

Indicator S16 contributes to WEV3 and assesses the influence on a wetland's hydrological state of linear transport infrastructure (roads and railways) in the 1 km buffer zone.

The impacts of linear transport infrastructure on the hydrology of freshwater ecosystems have been considered from various perspectives, under diverse conceptual frameworks and at different geographic scales (e.g. Jones et al. 2000; Coffin 2007; Blanton and Marcus 2009; Duniway and Herrick 2011 and 2013; Raiter et al. 2018).

Linear transport infrastructure can impact natural surface flows by impeding, diverting and concentrating water movement, thereby decreasing or increasing runoff into floodplain wetlands. Consequently, the natural movements of sediment, nutrients, wildlife, propagules and other waterborne ecosystem components into and through wetlands, and across floodplains, are changed. The wetland environmental values of biotic integrity (WEV1), local physical integrity (WEV2), and aquatic and ecological connectivity (WEV4) can each become impacted in turn, and overall wetland state degraded through a range of complex interactions.

Elevated embankments or levees constructed on floodplains, including those built to support roads and railway lines, redirect the natural overland flow of water and interfere with hydrological connectivity as well as the natural movement of sediment, nutrients and some aquatic organisms (Steinfeld and Kingsford 2011; Trombulak and Frissell 2001). Road crossings often act as barriers to the movement of aquatic species, including fish (Trombulak and Frissell 2001). Terraforming to create road or rail embankments can exacerbate soil erosion, concentrating and directing sediment-laden runoff into waterways and wetlands, increasing turbidity and degrading habitat for aquatic biota (Trombulak and Frissell 2001). Road and rail corridors are often subject to vegetation and soil disturbance, with verges and drainage ditches providing opportunities for weed establishment and further spread via wind, water and moving vehicles (Trombulak and Frissell 2001; Ehrenfeld 2008).

Constructed embankments and drainage ditches can exacerbate the natural tendency for the floodplain to become dry away from channels, preventing overbank flows spreading laterally across the floodplain and isolating some wetlands from fluvial processes. Steinfeld and Kingsford (2013) found that, in the Macquarie Marshes, the effects of upstream river regulation combined with local floodplain earthworks (including linear features) have contributed to significant changes in vegetation on the Macquarie floodplain, with flood-dependent communities displaying successional shifts toward more xeric terrestrial vegetation types. For example, areas with a higher density of floodplain earthworks have experienced greater hydrological disconnection than areas with lower density of earthworks (Steinfeld and Kingsford 2013).

In other situations, earthworks can constrain flows, as outlined in relation to Indicator S9, potentially changing the hydroperiod, with effects on biota persisting until the barrier to flow is removed (Gergel 2002). Changes in biological structure (driven predominantly by plant assemblages) and biogeochemical functioning (driven primarily by the storage of carbon in wetland soils) can remain for many decades even after flows are restored (Moreno-Mateos et al. 2012).

Raiter et al. (2018) examined the extent to which linear transport infrastructure in the form of tracks, roads and railways altered surface hydrology at the regional scale. They tested two hypotheses: that indications of altered surface water movement are associated with linear infrastructure and that the probability of impacts increased with the level of engineering of the infrastructure. In other words, that highly engineered structures such as railways would have a greater impact on hydrological functioning than dirt tracks, with intermediate impacts for intermediate levels of engineering. Both hypotheses were upheld and they concluded that alterations in surface water movement associated with linear infrastructure are 'numerous and pervasive,' with interception of natural overland and nearsurface flows and artificial changes to 'site-scale moisture regimes.' In addition, they found that linear infrastructure frequently triggers or exacerbates soil erosion. Furthermore, infrastructure impacts were found to interact with the level of grazing pressure on the areas surrounding the infrastructure, as well as with other variables such as soil type, climate, topography and vegetation.

Wetland Environmental Value (WEV) 4 – Connectivity

WEV4 has two state connectivity indicators each addressing structural, biological and potential connectivity. One is a landscape scale indicator and the other a local scale, buffer-zone indicator. Landscape connectivity is generally defined as a combination of structural connectivity (the arrangement of habitats in the landscape) and biological connectivity (how mobility and responses of species to the landscape influence their patterns of movement between habitats), whereas potential connectivity is the possibility for connectivity to occur based on the existence of the physical environment required for connectivity processes to take place (DEHP 2012, Morris 2012; DELWP 2015). In the context of these indicators, connectivity considers the natural interaction of a wetland with other ecosystems, including other wetlands.

The importance of connectivity to wetland condition has been highlighted in this review several times. From an ecological perspective, connectivity broadly refers to biological or physical connections among plants, animals and habitats within the landscape, giving plants and animals the ability to move through space and time (Tischendorf and Fahrig 2000; Morris 2012). Ecological connectivity has been defined as *the unimpeded movement of species and the flow of natural processes that sustain life on Earth* (CMS 2020), and can be considered as the flow of energy, materials or organisms between habitats and the way environmental processes (e.g. dispersal, feeding, breeding) are spatially and temporally facilitated at local and landscape scales. Connectivity contributes to maintaining physical, chemical and biological diversity, as well as the evolution of species and ecosystems, including wetland dependent species (e.g. Jenkins and Boulton 2003; Atum et al. 2008; Sheaves 2009; Hansen et al. 2010; Steinfeld and Kingsford 2013; Adame 2019; Sahlean et al. 2020). As such, connectivity is species and context dependent, being defined by the needs of the organisms living within, and moving through, habitats. Connectivity occurs both within and outside the water environment (Marczak et al. 2010; DEHP 2012; Rudnick et al. 2012; Jyväsjärvi et al. 2020), with movement between habitat patches occurring in response to ecological processes operating at multiple spatial scales (Cohen et al. 2016).

Wetland connectivity processes are not confined to hydrologic flow paths but operate across and between environmental realms i.e. across marine, freshwater and terrestrial ecosystems (Beger et al. 2010). For species that move over land or through air (e.g. amphibians, reptiles, birds, plant propagules), connectivity is enabled or constrained by species traits, terrestrial landscape characteristics, natural disturbances, land use types and human impacts (e.g. Gibbons 2003; Boughton et al. 2010) as well as the amount of habitat available.

Indicator S13 – Landscape vegetation connectivity

Indicator S13 contributes to WEV4 and provides an area-weighted average connectivity score for wetlands by assessing three fundamental ecologically important elements of connectivity: the extent of the remaining natural area (remnant vegetation and areas of regrowth), its condition (the core patch, i.e. the natural area minus edge, is used as a surrogate) and spatial separation (distance between natural areas).

The connectivity of and interactions among freshwater and terrestrial ecosystems is best understood when wetlands are considered integral parts of a mosaic of habitats within a terrestrial matrix (Mushet et al. 2019). From this perspective of interdependence, wetlands and terrestrial systems are part of a continuum, each playing an important role in functioning landscapes (Mushet et al. 2019), in which the biological function and role of the landscape matrix depends both on its composition and associated species (Meiklejohn et al. 2010). For example, many wetland animals spend considerable time in terrestrial habitats, so these habitats are important at the population, community and ecosystem levels (Gibbons 2003). Also, animals that do not reside (permanently) in, but are dependent on, wetlands include migrant species and those indirectly dependent on wetland biota such as canopy insects (Gopal 2009).

Mosaics of wetland habitats can be linked by corridors of terrestrial vegetation. Such corridors often join habitats with intact natural areas, together forming ecological networks in otherwise fragmented landscapes (Hilty et al.

2020). Corridors of native terrestrial vegetation (grassland, woodland and forest) can facilitate the movement of organisms and processes between areas of intact habitat, including wetlands, and contribute to maintaining ecological processes. They support processes such as plant dispersal, and the migration, dispersal or commuting of fauna (Meiklejohn et al. 2010; Sahlean et al. 2020) including reptile and amphibian species (Semlitsch and Bodie 2003; Roe and Georges 2007; Sahlean et al. 2020), dragonflies (Kietzka et al. 2015, 2021), soil fauna (Portela et al. 2019) and mammals (Paetkau et al. 2009; Carthew et al. 2013).

'Potential connectivity' is suggested by presence of structural connectivity linking native vegetation, buffer areas, corridors and larger habitat areas across the landscape. Furthermore, substantial areas of native terrestrial vegetation are needed to ensure structural connectivity (Gibbons 2003; Semlitsch and Bodie 2003; Marczak et al. 2010). Where corridors and patches are fragmented, opportunities for spatial and temporal patterns of movement within the wider landscape are limited, thus structural and biological connectivity are constrained, impacting the overall condition of wetlands. As such, the continuity of native vegetation via vegetation corridors and proximal patches of native vegetation is critical within fragmented landscapes for supporting movement of materials and species across and between habitats, and is integral to the realisation of potential connectivity processes and ultimately wetland biodiversity and its ecological condition (Brooks and Wardrop 2014; Stapanian et al. 2018).

Indicator S14 – Native vegetation in the 200 m buffer

Indicator S14 contributes to WEV4 and assesses the percentage area of native vegetation within a wetland's 200 m buffer zone.

Terrestrial areas in wetland buffer zones are a fundamental contributor to wetland environmental values (Roe and Georges 2007; DERM 2011; DENR 2018). Buffer zones comprising native vegetation provide critical habitat for a wide range of fauna living in and around wetlands (Naiman and Décamps 1997; Fischer and Fischenich 2000; Semlitsch and Bodie 2003). As outlined above, vegetated buffer zones also support plant and animal movements such as those associated with dispersal, feeding, basking, nesting, hibernation, migration and sheltering (Boyd 2001; Semlitsch and Bodie 2003; Roe and Georges 2007; Morris 2012) along with other connectivity functions such as providing detrital inputs to wetlands (Fischer and Fischenich 2000).

Methods for determining the structural and functional connectivity needs of species and for establishing preferred connectivity objectives and processes within buffer areas and the landscape can be complex and time consuming. However, 'potential connectivity' can be indicated by the presence of vegetation that can provide a physical environment for connectivity processes to occur (DEHP 2012). As such, measuring the extent of a native vegetation within a prescribed buffer area is a practical, indirect spatial indicator of connectivity processes.

Depending on species needs and management objectives, optimal buffers for supporting connectivity processes may be a few metres to several kilometres wide (Fischer and Fischenich 2000; DEWHA 2009a,b; Marczak et al. 2010; DEHP 2012; Cole et al. 2020; Jyväsjärvi et al. 2020). Wider vegetation buffers have higher interior to edge ratios and are generally considered more effective (Fischer and Fischenich 2000; Cole et al. 2020) because they allow species movement and support higher densities or higher populations of, for example, reptiles, invertebrates and larger animals (Semlitsch and Bodie 2003; Steen et al. 2012; Brooks and Wardrop 2014; Raitif et al. 2019) and greater biodiversity more generally than narrower vegetation buffers (Marczak et al. 2010; Ma 2016; Cole et al. 2020). Although fixed buffers contain inherent weaknesses in terms of capturing some connectivity functions, a 200 m native vegetation buffer is recognised as supporting wetland and species biodiversity (Hansen et al. 2010), maintaining a wetland's condition and integrity and, depending on the species involved, potential and actual connectivity functions at the local scale (Boyd 2001; Semlitsch and Bodie 2003; Marczak et al. 2010; DENR 2018).

Conclusion

The above review has provided an evidence base, collated and synthesised from the environmental, ecological and resource management literature, to support hypothesised connections between each of the WT indices and indicators and the condition of freshwater wetlands in the GBRCA (Figure 1).

Such evidence may be used to guide studies seeking to validate the efficacy of indirect indicators used for establishing pressure and state conditions, and to explore and identify specific causal links between *particular* pressures and their correlated state variables.

Overall, the review has confirmed the strong support in the literature for each of the WT indices and indicators and the theorised causal links between anthropogenic pressures on freshwater wetlands and the state of the WEVs that they measure. As such, the construct validity of the WT indices and indicators is supported.

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