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Effects of dredging-related pressures on critical ecological processes for organisms other than fish or coral.

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WAMSI Dredging Science Node

Report

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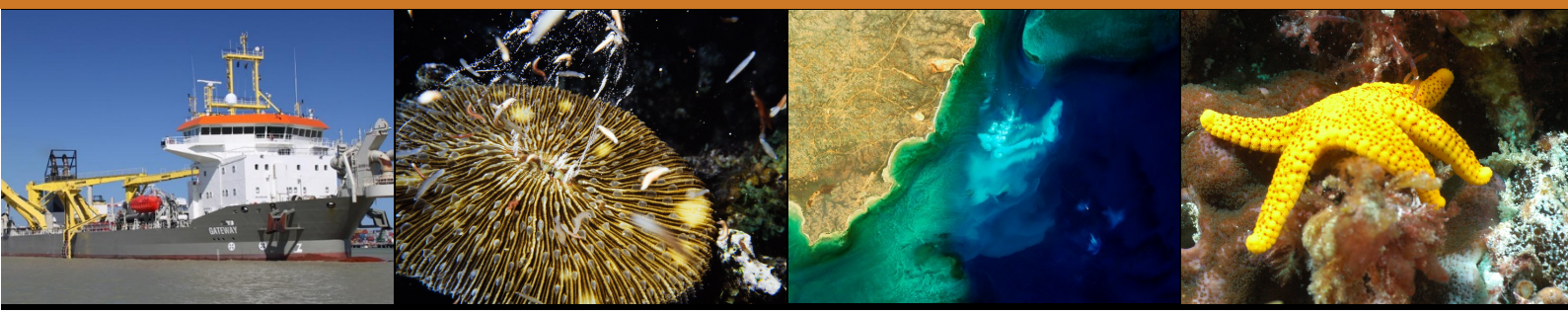

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WAMSI Dredging Science Node

The WAMSI Dredging Science Node is a strategic research initiative that evolved in response to uncertainties in the environmental impact assessment and management of large-scale dredging operations and coastal infrastructure developments. Its goal is to enhance capacity within government and the private sector to predict and manage the environmental impacts of dredging in Western Australia, delivered through a combination of reviews, field studies, laboratory experimentation, relationship testing and development of standardised protocols and guidance for impact prediction, monitoring and management.

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This remarkable **collaboration between industry, government and research** extends beyond the classical funder-provider model. End-users of science in regulator and conservation agencies, and consultant and industry groups are actively involved in the governance of the node, to ensure ongoing focus on applicable science and converting the outputs into fit-for-purpose and usable products. The governance structure includes clear delineation between end-user focussed scoping and the arms-length research activity to ensure it is independent, unbiased and defensible.

And critically, the trusted across-sector collaboration developed through the WAMSI model has allowed the sharing of hundreds of millions of dollars worth of environmental monitoring data, much of it collected by environmental consultants on behalf of industry. By providing access to this usually **confidential data,** the **Industry Partners** are substantially enhancing WAMSI researchers' ability to determine the real-world impacts of dredging projects, and how they can best be managed. Rio Tinto's voluntary data contribution is particularly noteworthy, as it is not one of the funding contributors to the Node.

Funding and critical data **Critical data**

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Front cover images (L-R)

Image 1: Trailing Suction Hopper Dredge *Gateway* in operation during the Fremantle Port Inner Harbour and Channel Deepening Project. (Source: OEPA)

Image 2: Many organisms including polychaete worms, undergo spawning events at the same time as corals. Cessation of dredging activities during mass coral spawning events will afford protection to a range of other marine organisms that have similar reproductive patterns. (Source: Ray Masini)

Image 3: Dredge plume at Barrow Island. Image produced with data from the Japan Aerospace Exploration Agency (JAXA) Advanced Land Observing Satellite (ALOS) taken on 29 August 2010.

Image 4: *Fromia* has been observed spawning at the same time as the mass coral spawning event at the Abrolhos Islands. (Source: Graham Edgar)

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Executive Summary

In November 2013 a workshop was held at CSIRO Floreat, which brought together national and international marine scientists. The workshop addressed two primary objectives:

- identify the timing of critical ecological processes in tropical and temperate ecosystems with a focus on non-coral and non-fish biota (seagrass, seaweed, sponges, ascidians, bryozoans, molluscs, echinoderms, crustaceans and non-coral cnidarians); and
- identify environmental windows for critical ecological processes identified in Objective 1. This will be achieved by compiling information on the timing of reproduction, release of propagules and recruitment for these organisms, as well as the temporal and spatial scales of reproduction and recruitment events.

During Workshop 1 a conceptual diagram was developed to illustrate and guide the decision process behind the selection of environmental windows (EWs) (see Figure 1). The life histories of the biota investigated were then identified and listed in detailed tables with specific reference to potential effects of dredging at each life history stage.

In September 2014 a second workshop was held at CSIRO Floreat. This workshop used the knowledge generated during Workshop I to address the remaining objectives:

- identify potentially critical periods and locations when mitigating scheduling and processes (EWs) could be employed to reduce the impact of dredging on non-coral and non-fish biota;
- review the state of knowledge regarding potential effects of dredge-related sediments and other dredge-related pressures on these key ecological processes; and
- identify the potential for invasive species to become established.

During Workshop II, the information within the life history tables was used to develop criteria for the assessment of vulnerability to dredging for marine invertebrates, seagrasses and macroalgae (Table 1), and this was used to assess the vulnerability for major Western Australian genera on an annual basis in order to identify EWs for dredging (Table 2, Figure 2).

Dredging activities may have drastic impacts on marine organisms, particularly the benthos. EWs, or the cessation of dredging during ecologically sensitive periods, can be an effective management tool if they are set properly. In addition to an understanding of environmental conditions, this requires location-specific knowledge of the timing of sensitive periods in the life histories of the key or dominant habitat forming organisms present.

The selection of effective EWs is highly dependent on the particular habitat and species present. These may be highly diverse, with correspondingly diverse life history characteristics and variable vulnerabilities to disturbance. Thus, the first step in the selection of EWs for dredging is to assess the ecological, social and economic 'value' of the species present in order to prioritise protection. Finally, the vulnerability of these species is assessed based on their life history characteristics and sensitivity to environmental change.

Marine invertebrates can play important roles in the habitats in which they occur. The filter feeders, in particular, are a highly diverse and ecologically important group, providing food and shelter for other sessile and mobile organisms. These can also be of great economic importance. For example, sponges have been used for the production of chemicals for biomedical research. As such, in habitats such as temperate reefs which are dominated by sponges and other filter feeders, these should be considered when making protection and management decisions.

Habitat forming primary producer taxa such as seagrasses and macroalgae should also take priority for protection and management. Seagrass meadows are highly important habitats in shallow coastal and estuarine ecosystems. They provide food, shelter and other ecological services to many ecologically and commercially important marine organisms and are amongst the most productive aquatic communities. Similarly, macroalgal beds are extremely ecologically important in most shallow temperate marine ecosystems, supporting diverse communities of fish and invertebrates.

In Western Australia (WA), there is the opportunity to improve and refine the use of EWs by identifying and understanding how dredging may impact key ecological processes in nearshore marine ecosystems. In WA, it is known that many marine organisms exhibit an increased vulnerability to disturbance during the late spring to early autumn period (Oct. – April) due to the timing of sensitive life history periods (periods of reproduction and recruitment), such that winter is a period of the year when dredging would pose the lowest risk to critical life cycle processes for a number of taxa. However, this does not hold true for ephemeral seagrasses. Furthermore, local information on potentially critical periods and detailed knowledge of life history characteristics are missing for many dominant WA species of invertebrates, seagrasses and macroalgae. Thus, there is a need to improve our knowledge of sensitive periods (reproductive periods, in particular) in the life histories of many WA marine species by undertaking a series of basic biological studies. We have also explored Dynamic Bayesian Networks (DBN) as an adaptive tool to assist in decision-making around EWs, investigating the effects of the timing and duration of dredging on ephemeral and persistent seagrasses in this pilot study. From the results obtained we recommend a full detailed study using DBN-type approaches be undertaken on a range of biota.

Considerations for predicting and managing the impacts of dredging

Collation of Information

Environmental windows (EW) are ecologically sensitive periods that, when known, can be used to inform dredging management decisions and minimize the risk of impacts on biota. EWs require location-specific knowledge of the timing of sensitive periods in the life histories of the organisms present. With this information, dredging activities can be planned and managed to avoid these periods leading to reduced environmental impacts and risk. The timing of EW will depend on the natural physical environment, as well as life history characteristics of vulnerable organisms in the community.

The known life histories of marine invertebrates (e.g. ascidians, bryozoans, sponges, molluscs, echinoderms, crustaceans and non-coral cnidarians) seagrasses and macroalgae have been collated and presented in the form of detailed tables with specific reference to potential effects of dredging at each life history stage.

The information in these tables provides a basis for identifying potential EWs and evaluating the degree of confidence that can be placed upon them.

The key considerations for invertebrates, seagrasses and macroalgae, and for reducing the likelihood of establishment of invasive species, are summarised below.

Invertebrates

Morphology plays a critical role in determining sensitivity to dredging pressures such as sedimentation: for example upright morphologies are generally more resistant to burial than encrusting forms. Similarly, motility is important with mobile invertebrates generally considered less vulnerable than sessile taxa to sedimentation, as they are able to re-orientate themselves or move to areas with less sediment build up.

A species' reproductive strategy, reproductive season and developmental strategy are major factors contributing to its vulnerability. For example, organisms which have a single reproductive episode in their life-cycles, would be expected to be more vulnerable to a dredging event compared to organisms which may reproduce multiple times in a lifecycle. Similarly, the effects of dredging during reproductive periods would be expected to be more detrimental for invertebrates with a discrete annual spawning period compared to those with multiple protracted spawning events occurring throughout the year.

Developmental strategy is also important. Brooding invertebrate species, with a limited capacity for dispersal are generally more vulnerable than those with planktonic larval stages that may facilitate the colonisation of new, undisturbed habitats. It is noted that many larval invertebrate species may have difficulty attaching to substrata covered in a layer of fine sediment.

Given this information there is a higher likelihood of a significant negative effect of dredging operations if they

are carried out during periods of larval release, settlement and recruitment.

Although limited, there is some evidence that a range of invertebrate taxa spawn synchronously with corals suggesting that autumn would be a period of particular importance in the life cycle of a range of tropical marine invertebrate species in north-western Western Australia.

On this basis, environmental windows established to reduce dredging related turbidity generation around the neap tide periods in autumn (e.g. to protect corals) would likely offer sensitive life stages of some important coral reef taxa at least some respite from turbidity-related stress in north-western Western Australia. EWs for other important invertebrate taxa will depend on factors such as reproductive seasons and strategies.

Seagrass

Seagrassess can be grouped into three broad categories (i.e. colonising, persistent and opportunistic) that reflect their reproductive, dispersal and growth strategies. These classifications are useful for planning and management as they can be used to assess relative vulnerability (ability to resist and recover from disturbance) of a particular species based on its life history characteristics.

Colonising species (e.g. *Halophila* spp; *Halodule* spp) have short ramet turnover times, are quick to reach sexual maturity and allocate a significant amount of energy into sexual reproduction to produce seeds, usually resulting in the presence of a seed bank. Species within this group generally have a limited resistance to disturbance but have the ability to recover quickly. In the wet-tropical Kimberley region, the lifecycle of *Halophila decipiens* follows light availability in deeper water habitats, with seed dispersal during the light-poor wet season, and seedling growth, meadow development and gamete production occurring during the dry season when water clarity and associated light availability is high.

Dredging activities during the dry season in the Kimberley region would place the greatest pressure on this species as the plants rely on higher light levels to stimulate germination of the seed bank, growth and meadow development and gamete production.

The seasonal growth and reproductive pattern for colonising seagrasses in the Pilbara is spatially and temporally variable and no clear and generally applicable EW can be specified at this stage.

Persistent species (e.g. *Posidonia* spp) have long turnover times, can contain significant energy stores, are slow to reach sexual maturity and place less investment in sexual reproduction with seed banks rarely present. As such this group is more resistant to disturbance but takes longer to recover than colonising species.

The focus for management in temperate regions where these meadows dominate is to reduce pressure during the summer months to increase flowering and fruiting success and to allow carbohydrates to be generated and stored to support seagrass survival during winter.

Opportunistic species (e.g. *Amphibolis* spp; *Zostera* spp*)* share traits with species from both of the previous classifications, with the ability to colonise quickly, produce seeds and to recover from seed when necessary. In Western Australia, Amphibolis species flower during the Austral autumn, with gametogenesis occurring between May and October. The seed germinates on the adult plant and is released as a mature seedling between November and June and seedlings are present year round. Therefore, it is possible that dredging in the months leading up to flowering (i.e. during autumn) could reduce carbohydrate reserves and flowering.

Avoidance of dredging during the warmer months is likely to be beneficial for species in the Zostera genus, while avoidance of dredging during the Austral autumn will be beneficial for Amphibolis species.

Macroalgae

As with seagrasses, environmental windows for macroalgae should account for plant phenology, sensitive periods in the life history cycle (e.g. gametophyte vs. sporophyte stages for some macroalgae) as well as annual cycles in environmental conditions. Sargassum and kelp (*Ecklonia radiata*) are the dominant canopy forming algae in Western Australia.

Sargassum spp*:* In temperate Western Australia, it appears that the most common phenology is a spring-summer growth period, followed by reproduction in late summer, followed by senescence, though this may not apply to tropical populations.

Ecklonia radiata: Production of zoospores in temperate habitats is seasonal, primarily occurring from early summer to autumn (Dec–May), with a peak in April. Winter is the season of slowest growth, and significant thallus erosion and dislodgement due to storm conditions. Based on these factors it appears that dredging during winter would be the least detrimental for *E. radiata* communities.

Chlorophyta and Rhodophyta*:* The phenology of most green and red algae is unknown and generalities with respect to these groups cannot be made at this stage.

Based on a vulnerability assessment for major Western Australian macroalgal genera and the known timing of reproduction and recruitment for these groups in temperate waters of the region, dredging would pose the lowest risk during August–September, when neither of the major habitat forming macroalgae are undergoing reproduction or recruitment.

Invasive species

Dredging can provide essentially barren sites for colonization that are free from competition by native species. To mitigate against this advantage, dredging might be scheduled to coincide with natural reproduction and settlement by native species. While it is accepted that habitat modification via activities such as dredging can enhance the spread of invasive species, it is noted that the scheduling of dredging to coincide with reproduction is at odds with the underlying tenet of the advice presented here, (i.e. that such periods would be the most vulnerable phase of a species' life history) and hence the strong preference is to avoid introductions rather than attempting to manage dredging to restrict establishment.

Overview

Locally-relevant information on life history characteristics and ecologically sensitive periods that would inform management decisions is lacking in a range of species of invertebrates, seagrasses, and macroalgae that are known or likely to be ecologically significant in Western Australia. The gaps in knowledge are particularly evident in the Pilbara which lies between the highly seasonal wet tropics to the north and the cool temperate zone to the south.

In temperate WA waters, many marine organisms exhibit an increased vulnerability to disturbance during the late spring to early autumn period (Oct. – April) due to the timing of sensitive life history periods (periods of reproduction and recruitment), such that the winter months generally represent the period of the year when dredging is likely to pose the least risk to arrange of taxa.

In more northern waters there is evidence to suggest that autumn is a particularly important period in the life cycle of many marine invertebrates associated with coral reefs. Reducing dredging related pressures at this time (particularly around the time of the annual coral spawning) would afford a level of protection to a wide range of tropical coral reef invertebrate taxa. The wet season in the Kimberley is considered to be the period when dredging-related pressures are likely to be least detrimental to the opportunistic seagrass communities that form extensive meadows in deeper waters. Within the Pilbara region, the spatial and temporal patterns in phenology and reproduction in seagrass communities is less clear.

1 Introduction

1.1 Dredging and its impacts on the marine environment

Dredging is the excavation and relocation of sediment from an area to allow for the development of coastal infrastructure and is accomplished by a variety of different techniques (PIANC 2010). Dredging perturbs marine environments by (1) increasing suspended sediment concentrations, (2) increasing sediment deposition, and (3) increasing turbidity (water cloudiness), resulting in reductions in light and the burial of benthic communities (EPA 2016). As a result, dredging can have significant impacts on the marine environment, particularly the benthos, if not managed effectively (Rogers 1990, Johnson et al. 1991, Desprez 2000). In order to understand the vulnerability of benthic taxa to impacts of dredging it is important to first understand their life history characteristics and sensitive life history stages. Due to high spatial and temporal variability in the occurrence of ecologically critical periods (e.g. reproductive periods), life history characteristics must be investigated on a species- and location-specific basis. Unfortunately, this knowledge is limited for many regions.

1.2 Setting environmental windows to mitigate the effects of dredging

One management strategy for mitigation of the impacts of dredging on marine flora and fauna is the application of temporal restrictions on dredging activities. Environmental windows (EWs) are a management practice used to reduce the impacts of dredging activities on marine biota through temporal restrictions. Globally, EWs are defined as periods during which dredging and the disposal of dredged material should have limited impacts, whereas seasonal restrictions are periods when these activities should be prohibited (NRC 2002). It should be noted that the Western Australian definition of an EW is the opposite. Setting effective EWs requires local ecological and environmental knowledge. A discrete period such as a mass spawning event is an example of a predictable period during which a population may be particularly sensitive. This information can be incorporated into the management of dredge operations to mitigate the effect on a particular species (Suedel et al. 2008). In some parts of the USA, there are several restrictions imposed on dredging activities during spring and winter to protect various species of fish (Reine et al. 1998, Suedel et al. 2008). For example, in San Francisco Bay, EWs are implemented to protect the commercially and ecologically important Pacific herring, which enter the bay during reproduction (Suedel et al. 2008). In Western Australia, seasonal restrictions on dredging activities have been used to protect coral mass spawning events in many locations across the state (Simpson 1985, Simpson et al. 1991, Babcock et al. 1994, EPA 2011).

Ecosystem-based management strategies are the most effective tool for managing environmental change (McLeod & Leslie 2009). EWs can be applied in a broader context, addressing seasonal changes in the susceptibility of an ecosystem as a whole to dredging. In the River Elbe, upstream of Hamburg Harbour, Germany, oxygen content tends to drop to near-critical levels for fish species during summer due to limited planktonic oxygen production at this time. Because the disposal of dredge spoils would increase the rate of oxygen consumption in the river, this activity is restricted from April through to October (Burt & Wallingford 2002). Similarly, the wet-dry climate of northern Australia produces discrete periods of higher turbidity in coastal waters during the wet season (October–March). During such intervals, marine organisms have adaptive strategies for coping with natural reductions in light levels and increases in turbidity (Lanyon & Marsh 1995). As such, dredging operations may be best carried out during the wet season in this region, when turbidity levels are naturally higher, and restricted at the onset of the dry season, thereby avoiding an increase in turbidity levels outside of the natural range when organisms would be more susceptible (van Senden et al. 2013). Furthermore, management strategies that incorporate a species' known tolerance to the specific impacts of dredging (e.g. reduced light), may allow for the application of environmental thresholds such that dredging activities can continue at particular times of year with little environmental impact. For example, in Gladstone, Queensland, Australia, a light-based threshold using a rolling average was applied to protect seagrasses from the detrimental effects of sediment-related reductions in light levels during the growing season (July–December) (Chartrand et al. 2012). Combining a temporal restriction with a threshold approach, like that employed for

seagrasses in Queensland, is likely to be a lesser impost than a year-round threshold approach or a complete restriction on dredging activities for the growing season.

Using EWs as a management strategy is theoretically effective, but in practice has been difficult to implement (NRC 2002). EWs for dredging were initially established in the USA to protect periods of migration for economically important taxa (shellfish, fish, etc.); however, the policy behind this strategy was disorganised and inconsistent, resulting in inflated dredging costs (Suedel et al. 2008). Adhering to EW policy is a major management challenge as it can be expensive and complex. Indeed, the procedure for setting EWs has not followed a particular structure and has, at times, lacked scientific basis (NRC 2002).

1.3 Objectives

The objectives of Workshops I & II were as follows:

- identify the timing of critical ecological processes in tropical and temperate ecosystems with a focus on non-coral and non-fish biota (seagrasses, seaweeds, sponges, ascidians, bryozoans, molluscs, echinoderms, crustaceans and non-coral cnidarians);
- identify environmental windows for critical ecological processes identified in Objective 1. This will be achieved by compiling information on the timing of reproduction, release of propagules and recruitment for these organisms, as well as the temporal and spatial scales of reproduction and recruitment events;
- identify potentially critical periods (environmental windows) and locations when mitigating scheduling and processes could be employed to reduce the impact of dredging on non-coral and non-fish biota;
- review the state of knowledge regarding potential effects of dredge-related sediments and other dredge-related pressures on these key ecological processes; and
- identify the potential for invasive species to become established.

2 Materials and Methods

Vulnerability to dredging was assessed for life history characteristics (LHCs) on a taxon-specific basis. For the invertebrates, six key LHCs were identified and used to predict vulnerability to dredging. These included: feeding strategy, mobility, life span, reproductive strategy, reproductive season and developmental strategy. A vulnerability index (VI) was then developed in order to assign a vulnerability score to each ecologically important taxon based on its particular LHCs.

For the seagrasses and the macroalgae, sensitive periods in the life histories were identified and their response to decreases in light, burial and sedimentation was reviewed. Vulnerability to dredging was then predicted for major genera.

A conceptual model of the process involved in determining EWs for dredging for a particular taxon or sub-taxon is shown in Figure 1. The level of accuracy of model predictions is inversely proportional to the level of generalisation of life histories within each group, directly proportional to the accuracy in predicting the magnitude of dredging-related damage, and also depends on identifying feedback mechanisms between the dredge pressure and species' responses. This becomes more complex when the timing of life history stages are considered, as these sensitive ecological processes may differ between taxa within our groups. Thus, the impact of dredging may vary across species in the same group in a particular area, however, this is not generally the case with seagrasses and macroalgae, and generalisations can be made based on season, sea temperatures and light reaching the benthos.

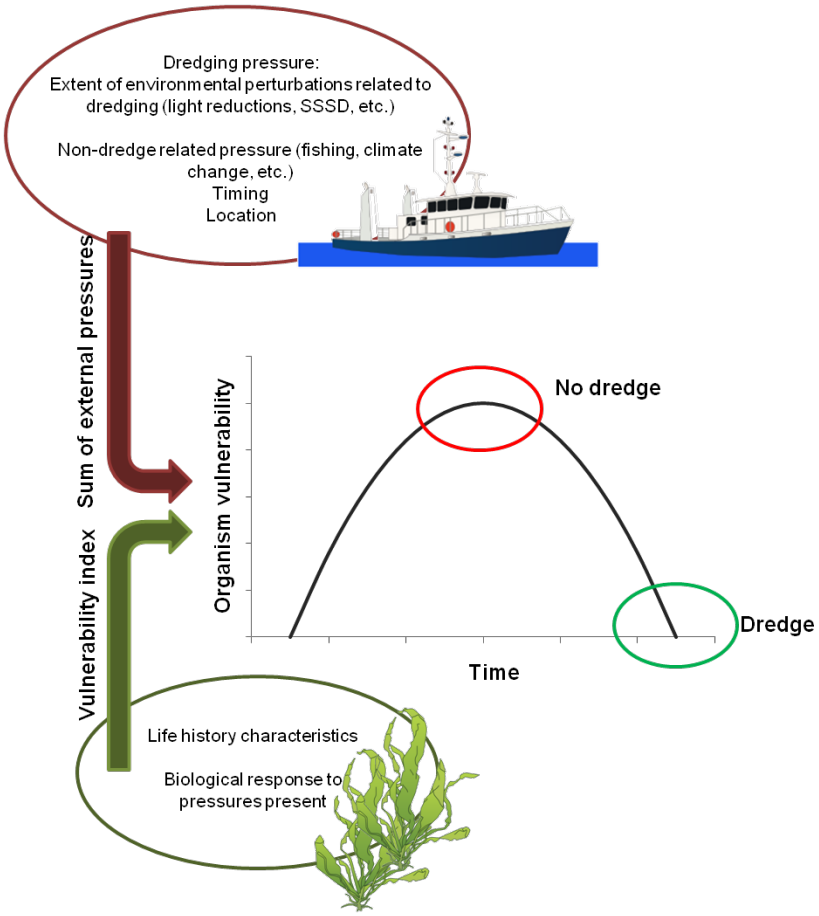

Figure 1. Input requirements for environmental window modelling. Once the structure of the model is known, the life history characteristics for a particular species must be identified in order to form a vulnerability index.

The life histories of non-coral and non-fish marine biota (seagrass, seaweed, sponges, ascidians, bryozoans, molluscs, echinoderms, crustaceans and non-coral cnidarians) were identified and listed in detailed tables with specific reference to potential effects of dredging at each life history stage in order to develop a species vulnerability index and to determine when EWs may exist on an annual basis.

During Workshop I we identified important LHCs for ecologically important marine taxa, and predicted the effect of dredging on them. During Workshop II we developed this information further by identifying representative Australian species for each of the relevant taxa and, through analysis of the literature, determined when the sensitive periods in their life cycles occur on an annual basis. The information within the resulting life history tables were used to build a decision making framework model for identifying appropriate EWs. In addition, Julian Caley, Paul Wu and colleagues have used the conceptual risk-model to develop a working Bayesian Network Risk-Based Model. During Workshop II, this model was validated and consensus and confidence levels were established for all aspects of the model. In addition, a range of scenarios that could be used for model validation were generated.

3 Results and Discussion

3.1 Invertebrates

Traits such as mobility, feeding mode, morphology and reproductive strategy contribute to the net vulnerability of a species to a dredging event (Essink 1999). Mobile invertebrates are generally less vulnerable than sessile taxa to sedimentation, as they are able to move to areas with less sediment build up or by physically removing

particles. Powilleit et al. (2009) measured mixed responses to sedimentation in the laboratory for Baltic Sea invertebrates, with survival rates of 0–33% depending on species and burial depth. Adult bivalves *Arctica islandica, Macoma balthica* and *Mya arenaria* and the polychaete *Nephtys hombergii* demonstrated a relatively high percentage of escape (restored contact with surface water) after burial in 32–41 cm. Some polychaetes (*Bylgides sarsi*) managed escape from 16 cm of sediment, while others (*Lagis koreni*) made no effort to migrate (Powilleit et al. 2009). Mobility alone does not indicate that these groups are resistant to dredging as they are still susceptible to several indirect effects of sedimentation. For example, juveniles of the blackfoot abalone (*Haliotis iris*) in New Zealand are not directly impacted by sedimentation, but have been observed to reorientate themselves during sedimentation events from a horizontal position underneath the cobbles (a predation refuge) to an upright position on the sides of the cobbles, increasing their vulnerability to predation (Chew et al. 2013). These examples highlight the importance of understanding individual species response to sedimentation on a location-specific basis as well as how ecological interactions may be modified under such conditions.

Sessile invertebrates are vulnerable to sedimentation because they are generally unable to reorientate themselves to mitigate a build-up of particulates. Some sessile taxa, including species of sponges and bivalves, have the capacity to filter out or to physically remove particulates, however this is metabolically costly and unsustainable (Gerrodette & Flechsig 1979, Cortés & Risk 1985, Aldridge et al. 1987). The impact of sedimentation on sessile invertebrates depends on a range of additional factors. Morphology plays a critical role since upright morphologies are generally more resistant to burial than encrusting forms. Indeed, studies on the sedimentation and burial of rocky sublittoral sponge communities have measured a decrease in morphological diversity with increased sedimentation (Carballo 2006). Similarly, sea whips and other gorgonian species in the Florida Gulf have been found to be relatively resistant to dredge-related sedimentation due to their morphology, which resists the build-up of sediment (Marszalek 1981). Diet and feeding mode are also important in driving species vulnerability to sedimentation. Sedimentation events can be particularly detrimental for suspension feeding organisms since suspended particles can be mistaken for food. In addition, the mechanical or abrasive action of suspended sediments may be harmful to suspension feeders, clogging their feeding apparatus and impairing respiratory and excretory function (Sherk 1972).

A species' reproductive strategy, reproductive season and developmental strategy are major factors contributing to its vulnerability. For example, semelparous organisms, which have single reproductive episode in a life-cycle, would be expected to be more vulnerable to a dredging event compared to iteroparous organisms, which may reproduce multiple times in a lifecycle. Similarly, the effects of dredging during reproductive periods would be expected to be more detrimental for invertebrates with a discrete annual spawning period compared to those with multiple protracted spawning events occurring throughout the year. In Western Australia, most species of scleractinian corals are known to spawn synchronously after sunset on an ebbing neap tide during a discrete and predictable annual window in autumn (Simpson 1985, Simpson et al. 1991). Species from other phyla have also been observed spawning in concert with the corals during these annual autumn spawning events. The most obvious is the polychaete worm (*Eunice* spp.) that releases a bright red epitoke, which is a reproductive segment carrying gametes that detaches from the rear of the worm. The epitoke is free swimming and possesses an eyespot. The epitoke is positively phototactic and as such will tend to head towards the brightest point, normally the sea surface. They will be attracted to other light sources such as underwater torches where they can form dense swarms containing many thousands of rapidly swimming epitokes. Polychaete spawning has been known for centuries in the Indo-west Pacific and Indonesia where the epitokes of the palalo worm (*Eunice viridis*) have been traditionally harvested for food. In WA this phenomenon has been observed to occur synchronously over 12 degrees of latitude from Dampier in the north (C. Simpson pers.com.), through Ningaloo Reef (R. Masini pers.com.) and as far south as Rottnest Island (K. Crane pers.com.) during studies on coral spawning. Echinoderms (sea stars and urchins) have also been observed spawning coincident with corals at Ningaloo (R. Masini pers.com.). The sea stars were typically seen on a local high-point of reef adopting a spawning posture, with body raised off the substrate, and releasing buoyant, red eggs into the water column. The urchins were observed releasing what appeared to be sperm. Less well known are the spawning events of species which are

not synchronised with scleractinians. Occasional observations which define spawning times of Western Australian marine species are made but these are rarely reported formally (e.g. Keesing et al. 2011a,b, Keesing and Irvine 2013),

Developmental strategy is also important. Brooding invertebrate species, with a limited capacity for dispersal are generally more vulnerable than those with planktonic larval stages that may facilitate the colonisation of new, undisturbed habitats. However, meroplanktonic species entering or remaining inside an area being dredged may be highly vulnerable to the mechanisms of dredging since high levels of sedimentation can inhibit larval settlement and recruitment. In the water column, bivalve and oyster larvae can tolerate suspended particulate matter (SPM) concentrations of up to 400–800 and 2200 mg L^{-1}, respectively (Wilber & Clarke 2001). However, once ready to settle, many larval invertebrate species may have difficulty attaching to substrata covered in a layer of fine sediment (Ray et al. 2005). Indeed, the deposition of sediment on mussel beds hinders settlement, attachment and survival of mussel larvae (Bender & Jensen 1992). In contrast, oyster larvae require clean, hard substratum for attachment, but can tolerate thin layers of sediment (up to 1mm). In the early stages of attachment, the deposition of fine sediments is likely to have a negative effect on recruit survival, whereas following this period oysters can tolerate sediment deposition of 2–3 mm, but >5 mm is likely to have negative effects (Wilber & Clarke 2001). Similarly, the Florida Keys spiny lobster, *Panulirus argus*, has reduced rates of settlement in heavily silted areas (Herrnkind et al. 1988). Fine sediments may also create a boundary layer for gas transfer, facilitating the formation of sulphides and creating anoxic conditions (Salomons 1985), which may inhibit the growth of attached organisms (Essink 1999). Given this information there is likely to be a significant negative effect of dredging operations if they are carried out during periods of larval release, settlement and recruitment such that identifying the timing of these ecologically sensitive periods on a species-specific basis is important for environmental window modelling for a particular location.

Table 1. Life history characteristics used to determine vulnerability to dredging for (A) invertebrates, (B) seagrasses and (C) macroalgae.

Group	**Characteristic**	**Vulnerability Score**		
		High	**Medium**	**Low**
Invertebrates				
	Feeding strategy	Autotrophs/filter feeders	Grazers/predators	Deposit feeders
	Movement	Sessile	Weakly mobile	Mobile
	Lifespan	Short-lived		Long-lived
	Reproductive strategy	Semelparous		Iteroparous
	Reproductive season	Discrete		Protracted
	Developmental strategy	Brooders	Lecitho- /planktotrophs	Asexual
Seagrasses				
	Growth rate	Slow-growing		Fast-growing
	Time to sexual maturity	Long		Short
	Turnover time	Slow		Fast
	Seed bank presence	Absent		Present
Macroalgae				
	Growth rate	Slow-growing		Fast-growing
	Lifespan	Longer-lived (years)		Shorter-lived (days- months)
	Reproductive strategy	Less complex (fewer stages)		More complex (more stages)

Table 2. Summary of known vulnerable periods for representative species of Western Australian invertebrates, seagrasses and macroalgae. For the invertebrates periods of spawning and reproduction, and for the seagrass and macroalgae periods of reproduction, recruitment and growth are shown. Species with unknown vulnerable periods are not shown.

Taxa	Descriptor	Representative species	J	F	M	A	M	J	J	A	S	O	N	D
Sponges	Encrusting	*Pione velans*												
	Encrusting with some autotrophs	*Chondrilla australiensis*												
		Lamellodysidea herbacea												
		Clinona spp.												
		Xestospongia testudinaria												
Ascidians	Solitary	*Pyura, Polycarpa* spp.												
		Didemnum spp.												
Bryozoans		*Bugula* spp.												
Molluscs	Gastropods (lecithotrophs/brooders)	*Notocypraea*												
		Zoila spp.												
		Austrocypraea												
	Gastropods (planktotrophs/brooders)	*Nerita albicilla*												
		Trochus histrio												
		Turbo bruneus												
	Bivalves	*Tridacna* spp.												
	Cephalopods	*Sepia apama*												
		Octopus maorum												
		Sepioteuthis australis												
	Chitons	*Acanthopleura gemmata*												
Echinoderms	Asteroids (broadcast spawners/planktotrophs)	*Linckia laevigata*												
	Ophiuroids (broadcast spawners/lecithotrophs)	*Ophionereis dubia*												
		Ophionereis semoni												
	Ophiuroids (broadcast spawners/planktotrophs)	*Ophiactis maculosa*												
		Ophiactis savignyi												
	Echinoids (broadcast spawners/lecithotrophs)	*Heliocidaris erythrogramma*												
	Echinoids (broadcast spawners/planktotrophs)	*Diadema savignyi*												
		Echinometra mathaei												
	Holothuroids (broadcast spawners, planktotrophs)	*Holothuria leucospilota*												
		Holothuria atra												
		Stichopus chloronatus												
Crustaceans	Crabs	*Portunus pelagicus*												
		Scylla serrata												
	Prawns	*Penaeus semisulcatus*												
Cnidaria	Soft corals	*Lobophytum crassum*												
Seagrasses	Temperate species	*Posidonia* spp.												
		Amphibolis spp.												
		Zostera spp.												
	Tropical species	*Thallassia* spp.												
		Enhalus spp.												
		Halophila spp.												
Macroalgae	Phaeophyta	*Sargassum* spp.												
		Ecklonia radiata												

	Representative species from WA		Representative genus elsewhere in Australia
	Representative genus from WA		Representative species overseas
	Representative species elsewhere in Australia		Representative genus overseas

Generalised summary of which invertebrate characteristics may be vulnerable to dredging for various life history stages is shown in Table 1(A). Detailed information on the LHCs of major invertebrate taxa is shown in Appendix 6.I. We have created sub-categories for each phylum, based on morphology and reproductive strategy, as these are major contributing factors to a species' vulnerability to sedimentation. This information was then used to assign a general vulnerability score to each taxon or taxon subset in Appendix 6.2.

Reproductive periods and the periods shortly thereafter generally represent extremely sensitive periods in the life histories of marine benthic invertebrates. A vulnerability index based on LHCs and timing of reproduction for known species of marine invertebrates in WA is shown in Appendix 6.2. Based on this information, vulnerable periods for WA invertebrates are shown in Table 2. Despite their importance this information is incomplete, as the timing of reproductive events for many of these species has not yet been investigated. Based on the available information, a potential EW for dredging on invertebrate dominated reefs in WA appears to be during the winter months (June–Sept., Figure 2), when the least number of known species are undergoing periods of reproduction and recruitment. However, the precise timing of the EW selected will be highly dependent on the particular invertebrate taxa present in any particular habitat and in north-western Western Australia, coral spawning is an accepted environmental window of importance. This is being dealt with separately (WAMSI Dredging Node Theme 7) and as there is some evidence that other invertebrates also spawn synchronously with corals (see above), autumn would be a period of particular importance in the life cycle of a range of tropical marine invertebrate species in north-western Western Australia. However there are also records of invertebrates in tropical Western Australia spawning outside this period (e.g. heart urchins in June (Keesing and Irvine 2013) and sea stars in November (*Protoreaster* spp. in November, Keesing pers. obs.) so there is a great need to establish a more synoptic picture of environmental windows in Western Australia built on new biological and ecological studies. However, in the absence this information, environmental windows established to reduce dredging related turbidity generation around the neap tide periods in autumn would likely offer sensitive life stages of some important taxa at least some respite from turbidity-related stress in north-western Western Australia.

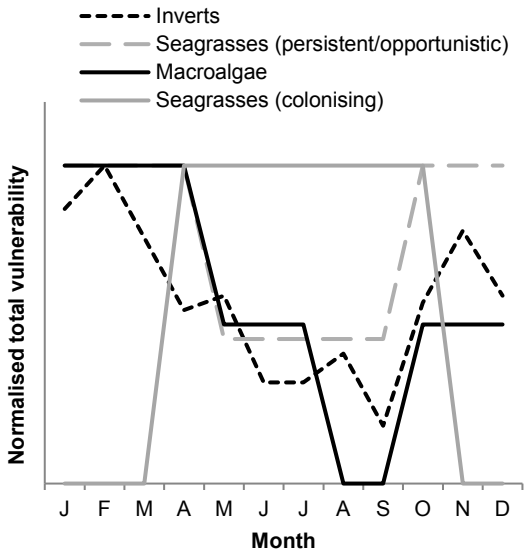

Figure 2. Normalised total annual vulnerability based on the timing of sensitive life history periods (Table 2) and vulnerability scores (Appendices 6.1, 6.5, 6.6) for representative species of invertebrates, seagrasses and macroalgae in Western Australia.

3.2 Seagrass

Seagrasses are highly sensitive to changes in water quality, sediment loading, and other inputs that accumulate as a result of the modification of watersheds and coastal water bodies (Dennison et al. 1993). Seagrasses are therefore useful for identifying critical environmental thresholds that may be triggered by dredge operations for other organisms. Given the widespread distribution and environmental/economic value of seagrass ecosystems (Orth et al. 2006), these organisms should be a priority for protection within dredging management practice

(Waycott et al. 2009). Seagrasses are affected by dredging in several ways. They are directly affected at the dredge and disposal sites, where they are often physically removed or buried, and indirectly affected by temporary reduction in dissolved oxygen, increase in pollutants and nutrients from contaminated sediments, or bathymetric changes which may sometimes occur with dredging activities (Erftemeijer et al. 2006). Most importantly, seagrasses are affected by the increase in turbidity levels, resulting in reductions in light available for photosynthesis, and by burial, which can result in significant negative effects on seagrass shoot density and leaf biomass, physiology and productivity (Erftemeijer et al. 2006).

The ability of seagrasses to resist and recover from disturbance such as a dredging event is species-specific and related to a number of life history traits (Table 1(B)). Recently, Kilminster et al. (2015) summarised seagrass vulnerability to disturbance by grouping species into three categories based on their life history characteristics: 1) Colonising species are defined by these authors as those with short ramet turnover times, that are quick to reach sexual maturity and with a high investment in sexual reproduction to produce seeds, usually resulting in the presence of a seed bank. Species within this group generally have a limited resistance to disturbance but have the ability to recover quickly; 2) Persistent species are defined as those with long turnover times, that are slow to reach sexual maturity and with less investment in sexual reproduction such that the presence of a seed bank is rare. Persistent species are more resistant to disturbance but take longer to recover than colonising species; and 3) Opportunistic species share traits with both of the previous classifications, with the ability to colonise quickly, produce seeds and to recover from seed when necessary. These classifications are useful for planning and management as they can be used to accurately assess the vulnerability (ability to resist and recover from disturbance) of a particular species based on its life history characteristics (Kilminster et al. 2015).

In addition to the life history characteristics that contribute to seagrass vulnerability to dredging, response to reductions in light is also an important consideration in the assessment of overall vulnerability for seagrasses (Appendix 6.3). Life history characteristics which may influence vulnerability to dredging are shown in Appendix 6.5, and detailed phenological information is given below for major Western Australian seagrass genera. Given that the timing of reproduction and recruitment, as well as sensitivity to reductions in light are highly species-specific (Table 2, Appendix 6.3), EWs selected for the protection of seagrass beds in this region will depend on the species present in a particular habitat.

3.2.1 Temperate seagrass species

Posidonia spp.

Posidonia species are grouped within the persistent classification by Kilminster et al. (2015) based on their life history characteristics and are thus generally more resistant but slower to recover from disturbance. These species are found in temperate and sub-tropical regions of Australia. Water quality and other environmental conditions follow a strong summer-winter seasonal cycle in these habitats, wherein light levels and temperature are higher during summer and levels of suspended sediment are greater in winter due to increases in riverine input and storm-driven re-suspension. The capacity of large-bodied, foundation species to survive short-term reductions in light levels after a dredging event is high, but after extensive periods of shading these species tend to experience significant loss of biomass and shoot density, with minimal recovery. Posidonia response and recovery following periods of reduced light, burial and sedimentation is species-specific and dependent on many additional factors such as the extent of light reduction, the duration of such events, depth, etc. (Table 3). Indeed, *P. oceanica* and *P. coriacea* have the lowest light requirements 7–8 % surface irradiance (SI) (Duarte 1991) and 8 %SI (Westphalen et al. 2004), respectively, while *P. sinuosa, P. australi*s and *P. angustifolia* have minimum light requirements of 7–24% SI (Duarte 1991, Westphalen et al. 2004), 10% SI (Fitzpatrick & Kirkman 1995), and 7–24% SI (Duarte 1991), respectively.

Table 3. Summary of *Posidonia* spp. responses to (A) light reduction and (B) burial. SI = surface irradiance.

***Posidonia* Species**	**Light level**	**Duration (days)**	**Physiological Response**	**Recovery time**	**Reference**
(A) Light reduction					
P. australis	Sub-lethal	90	Decrease in shoot density and biomass	Little recovery	(Fitzpatrick & Kirkman 1995)
P. sinuosa	0–10% SI	148	Decrease in shoot density and productivity	Little recovery	(Gordon et al. 1994)
P. sinuosa				3.5–4 years at 3–4 m depth, 5 years (7–8 m depth)	(Collier et al. 2009)
(B) Burial	Depth (cm)				
P. oceanica	5		Decreased biomass		(Erftemeijer et al. 2006)
P. oceanica	15	45	Total shoot loss		(Cabaço et al. 2008)
P. augustifolia		14	Total mortality		(Clarke 1987)
P. australis	> 15	50	50% mortality		(Cabaço et al. 2008)
P. sinuosa	> 15	50	50% mortality		(Cabaço et al. 2008)

In Western Australia Posidonia species generally flower from autumn through to spring and fruit over summer. Seeds are released continuously throughout summer and may float for up to a few days before the seed is deposited. Seedling establishment will therefore occur within a few days to weeks following seed release and will extend over summer into autumn. Posidonia seeds have significant reserves and seedlings can grow on those reserves for up to one year (Statton et al. 2013), such that seedling growth and establishment is continuous, with the previous year's recruits overlapping with new recruits. Furthermore, productivity during summer is likely to be important in determining flowering and fruiting success. Carbohydrate stores are typically much larger in summer than winter (Collier et al. 2009). Seagrass δ13C values are less depleted in summer, indicating less light limitation and higher photosynthetic demand for carbon, though this is not reflected in rates of Posidonia primary production (Collier et al. 2009). Seasonal differences in leaf extension rates have been observed, with faster growth during summer (Collier et al. 2007), though these may vary between depths, and specific growth rates (Walker & McComb 1988). Furthermore, root length and number have been shown to be greater during summer for *P. australis* and *P. sinuosa*; factors which were also affected by nutrient levels, but to a lesser extent than season (Hovey et al. 2012). In contrast, there are no apparent seasonal trends in meadow areal production (Collier et al. 2007).

Amphibolis spp.

Amphibolis is in the middle of the resilience and recovery spectrum for seagrasses, and is classified as opportunistic (Kilminster et al. 2015), which is consistent with its morphology and life history traits. As a moderately fast growing, longer lived genus, Amphibolis is generally more resilient to disturbance than smaller morphological forms with shorter life spans. *A. griffithii* meadows have been shown to recover from shading experiments mimicking dredging scenarios lasting for 3 months followed by a 10 month recovery period, despite biomass losses of up to 72%, however recovery was limited following longer periods of shading (6–9 months) (McMahon et al. 2011). There is also some evidence indicating that this genus is resilient to sedimentation and burial, with the growth rates unaffected following burial in 10 cm of aerobic sediment along the Adelaide coast

(Clarke 1987).

In Western Australia, Amphibolis species flower during the Austral autumn, between May and October. The seed germinates on the adult plant and is released as a mature seedling between November and June and seedlings are present year round. Therefore, it is possible that dredging in the months leading up to flowering (i.e. during Autumn) could reduce carbohydrate reserves and flowering. Amphibolis meadows are more productive during summer (Dec–Feb) and reduced but relatively constant for the remainder of the year (Walker & McComb 1988), such that reserves are likely to be established during summer. Furthermore, the frequency of rhizome branching is highest during spring and summer (Walker & McComb 1988, Carruthers 1994), such that the capacity to recolonise may be greater at those times. Finally, experimental studies have shown a higher sensitivity in *A. griffithii* to post-summer, relative to post-winter shading such that avoiding dredging activities during post-winter (spring) periods may be beneficial for this group.

Zostera spp.

Seagrass species within the genus Zostera are also classified as opportunistic (Kilminster et al. 2015). In temperate environments, *Zostera* spp. have shown limited resilience to burial (70–90% mortality under 2–4 cm sediment) (Mills & Fonseca 2003, Cabaço & Santos 2007), and this may be due to a lack of vertical rhizome (Cabaço et al. 2008). Indeed, large losses of *Z. tasmanica* in Adelaide, Australia (445 ha between 1965 and 1985), were attributed to sediment build up on the leaf surface (Clarke & Kirkman 1989).

In the tropics, losses of thousands of hectares of *Z. muelleri* in Moreton Bay (Queensland, Australia), were attributed to dredging and the related increase in turbidity levels (Kirkman 1978). Despite these large losses, this genus has also been shown to have a relatively high capacity for recovery. In the tropics, *Z. muelleri* recovered completely within two years following a flood-related loss of 95% loss of intertidal seagrasses in the Great Sandy Strait, Queensland, Australia. Recovery was facilitated by seed banks stored in the sediment (Campbell & McKenzie 2004). Thus, maintenance of seed banks is critical to the recovery of damaged seagrass beds and dredging operations timed after propagule release will facilitate natural re-growth from seed reserves.

In Western Australia, *Zostera tasmanica* reproductive structures have been observed in September and mature flowers have been observed during summer (Dec–Jan) (Kirkman 1999). Other authors have observed evidence for sexual reproduction in this species, but only during 1 of 3 years monitored (Campey et al. 2002). Due to the episodic sexual reproduction observed as well as the lack of a stored seed bank, these authors concluded that sexual reproduction is unlikely to contribute significantly to the maintenance of Zostera populations. In other regions the greatest rates of Zostera leaf and areal production have also been measured during summer and late spring (Victoria, Australia (Bulthuis 1983, Bulthuis & Woelkerling 1983)). Thus, EWs which avoid the warmer months are likely to still be beneficial for species in the Zostera genus.

3.2.2 Tropical seagrass species

Thalassia spp.

Seagrass species within the genus Thalassia are large-bodied and slow-growing and thus classified as persistent (Kilminster et al. 2015). They are negatively affected by sediment plumes and sediment suspended in the water column due to the reductions in light availability associated with turbidity. *T. testudinum* in Corpus Christi Bay, Texas, had 99% mortality after 490 days under 14% SI, and 100% mortality after 200 days under 5 %SI. Furthermore, reductions in leaf productivity, displayed as a narrowing of leaf blades, and reductions in rhizome and leaf carbohydrate carbon content (50 and 15% less than unshaded control, respectively) were measured (Lee & Dunton 1997). The authors concluded that indices such as shoot density, blade width, leaf growth, Chl. a:b and blade chlorophyll content may be important early indicators of chronic stress due to light reductions associated with dredging. Thalassia species generally have a moderate tolerance to burial, with 50% mortality observed for *T. testudinum* and *T. hemprichii* under 5 cm of sedimentation (Suchanek 1983). Seagrasses within this genus have been observed to recover relatively slowly following disturbance, taking several years to recover from vessel-related injury (3.5–4.1 years to heal propeller scars and up to 7.6 years for artificial cuts) (Dawes et al. 1997).

It is therefore critical that Thalassia meadows are not damaged beyond the threshold of recovery, as complete recovery and regrowth of a damaged meadow may take many decades.

Enhalus spp.

Enhalus species are persistent and large-bodied, which may contribute to their resistance to sedimentation (Waycott et al. 2007, Cabaço et al. 2008). Indeed, only 20% mortality of *E. acoroides* was observed after 10 months buried under 16 cm of sediment (Cabaço et al. 2008). To the best of our knowledge, Enhalus response to shading has not yet been studied; however these species are known to occur in highly turbid environments. Given the traits discussed above, as well as its rapid growth rate (>1 cm d^{-1}) and relatively wide depth distribution (0–8m) (Johnstone 1979), Enhalus species would be expected to be relatively resistant to the short-term impacts of dredging.

Cymodocea spp.

Seagrasses within the genus Cymodocea are relatively sensitive to burial but have been shown to recover from periods of increased turbidity if conditions return to normal. Sudden burial under 5 cm of sediment resulted in 90% mortality in *C. nodosa* after 35 days (Marba & Duarte 1994). However, *C. serrulata* were able to withstand burial under 4 cm for 27 days, but responded adversely to burial depths greater than 8 cm with large reductions in above and below ground biomass and shoot density (Ooi et al. 2011). Furthermore, plants with experimentally severed rhizomes were more adversely affected by burial than those with intact rhizomes, indicating that species within this genus may better withstand burial when clonal integration is maintained (Ooi et al. 2011). This is likely due to the sharing of resources between neighbouring individuals (ramets) in large seagrass genera such as Cymodocea (Marba et al. 2006).

Cymodocea species have shown a relatively high capacity to recover from periods of eutrophication and/or light deprivation if conditions improve. This is demonstrated in research by Garrido et al. (2013), which monitored the natural decline and subsequent recovery of a *C. nodosa* meadow over a 40 year period. These authors report a 49% decrease in seagrass cover in a Mediterranean lagoon from the early 1970's to the early 1990s, followed by a 42% increase from the early 1990s to 2013. Heavy rainfall events, dredging and eutrophication increased the turbidity and reduced available light levels in the system, which lead to the initial seagrass declines. Recovery was initiated by improved catchment management, termination of channel dredge maintenance and a short term decline in herbivorous sea urchin abundance (Garrido et al. 2013). A number of factors may contribute to Cymodocea population fluctuations, and therefore the background ecology and feedbacks present in a particular community must be considered if the impacts of dredging are to be accurately predicted.

Syringodium spp.

Seagrasses within the genus Syringodium have a relatively high capacity to recover after a burial event, although the effects of prolonged burial, sediment type, interactions between the effects of reduced light and burial, and the implications of these for seagrass recovery require further investigation. After experimental burial in the Phillipines, *S. isoetifolium* experienced an initial decline in biomass but subsequently recovered. Shallow burial (2 cm) stimulated a growth response, with a significant increase in the number of shoots after two months. As with *Cymodocea serrulata, S. isoetifolium* were able to withstand burial under 4 cm for 27 days, but responded adversely to burial depths greater than 8 cm with reductions in biomass and shoot density (Ooi et al. 2011). This species also benefitted from the presence of an intact rhizome, such that clonal integration is important for the persistence of Syringodium species following a burial event (Ooi et al. 2011).

Halodule spp.

Small-bodied Halodule species are more vulnerable to short term pulses of increased turbidity and sedimentation than larger-bodied climax species. *H. wrightii* can survive in conditions measured between 5–30 %SI depending on the depth, water colour and natural turbidity fluctuation (Erftemeijer et al. 2006). However, the capacity for

recovery after a loss is reasonably high in this genus, with documented recovery of 2000 ha of seagrass in Tampa Bay, Florida, USA, due to improved water conditions.

Sedimentation and burial (4–8 cm depth) of a mixed seagrass meadow that included *H. uninervis* resulted in reductions in shoot density and limited recovery after 10 months (Duarte et al. 1997). However, the same species has been shown to withstand burial under 4 cm for 27 days, with reductions in biomass and shoot density only measured in burial depths exceeding 8 cm (Ooi et al. 2011). Furthermore, clonal integration is less important for the recovery of this genus following burial than the larger tropical seagrasses (Ooi et al. 2011) due to their relatively small size and limited communication and resource sharing between ramets (Marba et al. 2006).

3.2.3 Temperate and tropical seagrass species

Halophila spp.

Seagrasses within the genus Halophila and other colonising species grow quickly from a stored seed bank and may therefore re-colonise dredged areas through seed dispersal (Kilminster et al. 2015). However, there is a trade-off between fast growth and reproduction, which results in a relatively low tolerance to prolonged periods of decreased light levels compared to more persistent species. Halophila species are able to physiologically and morphologically acclimate to reductions in available light due to their relatively small morphological form. Longstaff et al. (1999) found that *H. ovalis* showed acclimation potential to light levels below their minimum light requirements, but only for 3–5 days, after which growth rates were reduced. Recovery was possible for this species if light levels were restored within 9 days, but periods of low light exceeding 15 days had an exponentially greater risk of mortality, with 100% mortality occurring after 30 days (Longstaff et al. 1999).

H. ovalis has a relatively low tolerance to burial (Vermaat et al. 1997). However, despite their small size, *H. ovalis* were able to withstand burial under 4 cm for 27 days, while burial depths greater than 8 cm resulted in large reductions in biomass (Ooi et al. 2011). Despite this, the relatively fast growth rates and high rates of reproduction characteristic of *Halophila* spp. and other colonising species, decrease their vulnerability to disturbance (Kilminster et al. 2015). Thus, they can generally recover following sedimentation and burial if seed banks are present. Therefore, as with many of the previously discussed seagrass genera, it is useful to protect reproductive stages in the life history for this group.

For some Western Australian Halophila species, such as *H. decipiens*, cycling between active growth and dormant seed bank stages in the life history is triggered by environmental cues, such that periods exist when dredging activity may have little impact. For example, the natural light climate in the Kimberley region, Western Australia, involves fluctuations of low light (10–0 %SI) in the wet season (Nov–April), and higher light levels (20-2 %SI) during the dry season (May–Oct). The lifecycle *of H. decipiens* follows light availability, with dormant seed dispersal stages during the darker wet season, and seedling growth, meadow development and gamete production occurring during the lighter dry season. As such, vulnerability to sedimentation and reduced light is low during the dormant seed dispersal stage during the wet season, and presents an EW for low-impact dredging. Dredging activities would likely have major impacts on this species during the dry season in this region, when the plants rely on higher light levels to stimulate germination of the seed bank, meadow development and gamete production.

3.3 Macroalgae

As with seagrasses, environmental windows for macroalgae must account for plant phenology as well as annual cycles in environmental conditions, which may be highly site specific in some instances. In terms of phenology, sensitive periods in the life history cycle should be considered (e.g. gametophyte vs. sporophyte stages for some macroalgae). Together with temporal variability in environmental factors as discussed above, these can reveal periods during which dredging activities are likely to have greater impacts on a particular habitat. Vulnerability scores based on LHCs for major Western Australian macroalgal genera are shown in Appendix 6.6. Based on these and the timing of reproduction and recruitment for these groups (Table 2), an EW for dredging in this region exists in August–Sept, when neither of the major habitat-forming macroalgae are undergoing reproduction or

recruitment (Figure 2).

3.3.1 Phaeophyta

Sargassum spp.

Brown algae within the genus Sargassum are common and important features in benthic ecosystems around the world. It is thought that these species have an advantage in higher sediment environments due to their abundance in turbid, inshore reef habitats (e.g. on the Great Barrier Reef). Indeed, Schaffelke (1999) observed an increase in rates of Sargassum growth of up to 180% when particulate matter (PM) was present on the thallus surface, potentially due to the creation of a nutrient-rich boundary layer. It appears that this group is resistant to the negative effects of sedimentation if it is already established in a system, but observed increases in Sargassum abundance may not be directly related to the sedimentation event. For example, a sedimentation event triggering coral die off would reduce the pressures of competition on these algae. Similarly, declines in fish and invertebrate populations could lead to reductions in herbivory. In contrast, increased sedimentation levels in a fringing reef environment led to significantly decreased rates of recruitment, growth, survival and vegetative regeneration in *Sargassum microphyllum* (Umar et al. 1998). It has also been noted that successful settlement of kelps and other algal species on hard bottom substrata is inhibited by sediment, with a direct relationship between settlement success and the thickness of the sediment, organic enrichment and/or the presence of sulphides for some algal species (Chapman & Fletcher 2002). Thus, the effect of sedimentation on species within the Sargassum is variable. Due to the increased sensitivity of Phaeophyta species to sedimentation during reproductive and recruitment phases, it would be beneficial to avoid these periods for dredging.

The detailed phenology of most brown algae is either unknown or location-specific. In general, *Sargassum* spp. are most abundant during the warmest part of the year in temperate regions; but most abundant when temperatures are lowest in the tropics (De Wreede 1976). As such, an annual cycle of vegetative growth, reproduction and senescence in Sargassum is often reported, but its timing varies between locations. In Hong Kong, Ang (2006) noted peak reproduction for *Sargassum* spp. between November–February, whereas it was during February–March for other species. It was suggested that phenology was temperature dependent, with reproduction occurring during the colder months. However, in a similar study in the Philippines, Ang (1985) suggested that tides may be a critical factor in structuring the *phenological* patterns of two Sargassum populations, which experienced die-back during a period of prolonged exposure related to the lowest tide of the year.

In temperate southwest Australia, Kendrick and Walker (1994) observed reproduction of *Sargassum* spp. during late spring - summer (September – December at Rottnest Island). Earlier, Kendrick (1993) noted that the seasonal timing of reproduction in *S. spinuligerum* varied with location and even between subtidal and intertidal habitats within the same location. Given the spatial variation in Sargassum annual reproductive cycles, we recommend site-specific planning for management purposes. In temperate Western Australia, it appears that the most common phenology is a spring-summer growth period, followed by reproduction in late summer, followed by senescence, though this may not apply to tropical populations.

Ecklonia radiata

As with all kelps, Ecklonia has a heteromorphic life history wherein the large, conspicuous plant (the sporophyte) alternates with a small, filamentous gametophyte, which is the site of sexual reproduction. Survival and growth of the gametophyte is tolerant of high and variable temperatures, but plants derived from early season sporangia are more successful than those from the late season.

Production of zoospores by the Ecklonia sporophyte in temperate habitats is seasonal, primarily occurring from early summer to autumn (Dec–May), with a peak in April (Mohring et al. 2013a, Mohring et al. 2013b). Based on the assumption that the period leading up to sporangial production, release and then gametophyte growth would also be sensitive to perturbation, we suggest that winter dredging would be the least detrimental. Winter is also the season of slowest growth, and significant thallus erosion and dislodgement due to storm conditions.

Underlying juvenile sporophytes may also contribute to the formation of a new canopy following a canopy loss, but this is dependent on the timing of canopy removal, with late summer–autumn loss favouring faster recovery (Toohey & Kendrick 2007).

3.3.2 Chlorophyta

The effects of dredging and sedimentation on the Chlorophyta (green algae) are similar to that on the Phaeophytes. Low levels of sedimentation are unlikely to inhibit algal growth but may affect recruitment, survival and vegetative regeneration. Furthermore, invasive Chlorophytes in the Mediterranean Sea, such as *Caulerpa racemosa* (now *C. cylindracea*), may be more resistant than native species to sedimentation events, thus benefitting from such disturbances (Piazzi et al. 2005). Calcareous green algae within the genus Halimeda have a noted tolerance to lower light levels (Hillis-Colinvaux 1986) and may be more resistant than species with higher light requirements to the increases in turbidity associated with dredging.

As with the brown algae, the phenology of most green algae is unknown. In the Caribbean, Clifton and Clifton (1999) noted a broadly seasonal peak of reproductive activity in green algae that coincided with the annual shift from dry to wet season in Panama (Mar–June); a period of increased solar radiation. In Australia, Price (1989) recorded active growth of most species during autumn, winter and spring, whereas smaller groups were restricted to winter and spring, and others to summer. As such, generalities with respect to Chlorophyta phenology cannot be made without considerable further study.

3.3.3 Rhodophyta

The distribution of crustose coralline algae (CCA) on the Great Barrier Reef has shown strong links to the sedimentation environment. Near shore reefs exposed to higher SSD had a much lower abundance of CCA, and abundance increased from the middle to the outer shelf with increases in visibility, reef slope and a decrease in sedimentation (Fabricius & De'Ath 2001). Despite distribution patterns suggesting that CCA are sensitive to sedimentation, this group has adapted a mechanism by which it can survive long periods of burial. Following burial, CCA are able to slough off epithelial cells such that underlying tissue can survive after the sediment is removed (Keats et al. 1997). Despite their resistance to the negative effects of burial, CCA are sensitive to the reductions in light associated with sedimentation (Riul et al. 2008). Furthermore, as with many other marine organisms, the CCA are more sensitive to sedimentation during certain periods of their lifecycle such as during recruitment. CCA are extremely important in the habitats in which they occur, contributing to carbonate accretion, structural complexity and facilitating the settlement and recruitment of many other taxa. As such, their response to sedimentation and burial will have major ecological ramifications on a community wide scale. In contrast, foliose Rhodophyte species are relatively tolerant to reductions in light. For example, the shade-adapted red alga *Anotrichium crinitum* has minimum light requirements of 1.49–2.25 µmol photons $m^{-2}s^{-1}$ and 0.12–0.19 mol photons $m^{-2}d^{-1}$ for the initiation of photosynthesis and growth, respectively. This group can also tolerate sub-optimal light conditions for up to five days without losing biomass (Pritchard et al. 2013).

There have been numerous phenological studies on the Rhodophyta, but considerable variation in the periodicity of growth and reproduction has been observed (Price 1989), such that no general pattern is evident. Maggs and Guiry (1987) suggested that temperature, photoperiod, light quality and irradiance are the most important environmental factors regulating macroalgal phenology, although temporal variation in nutrient levels, grazing pressure, wave action and sand scour may also be important in some communities. Rhodophyte life history may also play a role. Indeed, species with heteromorphic life histories including crustose or filamentous phases are often prevalent on mobile substrata and are able to withstand severe conditions, although existing studies are limited.

3.4 Invasive Species

Dredging activities have the potential to introduce marine pests or to encourage the spread of established marine pests. There is the potential for translocation of pest species attached to vessel hulls or in niche areas such as propellers, rudders, thrusters, stabilisers and other submerged areas of a vessel that provide attachment surfaces

and shelter during vessel movement (Ministry of Agriculture and Forestry 20 10). Non-indigenous pest species may also be released in adult of larval form with ballast water discharge. Pests pose a serious threat to native biodiversity (Wyatt et al. 2005). While many cause no apparent harm, others can cause diseases in humans and native species, disrupt ecosystems, damage fisheries and aquaculture activities and cause industrial problems such as fouling (Byrnes et al. 2007, Wells et al. 2009). A study by Huisman et al. (2008) reports 23 introduced pest species in the north of Western Australia from Geraldton to sites north of Broome. However, since then, there has been significant marine oil and gas development in the region, which has likely increased the risk of introduction of marine pests (Wells et al. 2009).

Thresher (1999) noted that establishment of pest species often follows environmental disturbance or the creation of new habitats, using the terrestrial example of weeds invading pristine forests along the edges of new roads as a comparable event. He suggested that dredging can provide sites for colonization free from the competition of native species, and to combat this dredging might be scheduled to coincide with natural reproduction and settlement by native species, or by 'seeding' disturbed areas with 'invader-resistant native species'. The scheduling of dredging to coincide with reproduction would appear to be at odds with the underlying tenet of much of this workshop, i.e. that such periods would be the most vulnerable phase of a species' life history. While it is accepted that dredging can undoubtedly enhance the spread of invasive species, this is predicated on the presence of source populations to 'seed' the disturbed area, which must be assessed on a case-by-case basis.

Dredging might also enhance the survival and growth of pest species by increasing the availability of nutrients previously bound in the substratum, or by placing stress on the native species and thereby enhancing the pest species' competitive advantage (Reise et al. 2006). Pest species generally thrive in disturbed areas (Galil 2000). Again, this assumes the prior presence of the pest. Nevertheless, the objectives of the current workshop are to protect native species by reducing stress, thereby increasing resilience to any real or potential threat posed by introduced or pest species.

Initial pest incursions might also be linked to the international or national movement of dredges and associated shipping, either by hull-fouling or ballast water discharge (Campbell & Hewitt 2011). However, these incursions are not a direct result of dredging *per se*, and are dealt with by existing quarantine and inspection protocols.

4 Discussion and conclusions

4.1 Seagrass vulnerability model

The effect of dredging operations on light levels reaching the benthos (% SI) should be the major factor considered in the development of environmental windows for the protection of seagrasses. % SI is ecologically relevant and is easily measured using light meters in the field. If it is possible to model the existing environment, accounting for additional factors affecting light attenuation (water colour, depth, natural suspended sediment regimes, etc.), then % SI reaching the benthos can be predicted for particular areas. Using existing knowledge of species-specific % SI thresholds we can predict the survival of existing meadows under the modelled conditions. Knowledge of natural turbidity regimes and species life histories will allow for the identification of periods during which dredging will have the lowest impacts (i.e. when will artificial increases in turbidity and sedimentation mirror the natural turbidity levels), as well as periods when seagrasses will be particularly susceptible to dredging. If losses are unavoidable (directly at the dredge site), it is necessary to identify the capacity for seagrass recovery, facilitated by seed banks or surviving populations via vegetative regeneration.

A Dynamic Bayesian Network (DBN) model of seagrass vulnerability and its evolution over time is being developed. This model is designed to capture environmental and biological factors and their causal effects on key metrics of seagrass meadow health including aerial extent, shoot density and biomass. Figure 3 provides a high level overview of the model showing links between different factors and vulnerability. The relationships between the factors are quantified through a combination of expert knowledge and available data and extend

to temporal dependencies and feedback loops. Thus, the model is capable of simulating or predicting variations in aerial extent, shoot density and biomass over time given the background situation for a meadow and the added pressures of dredging. A complete overview of how Bayesian Networks are used to model complex systems is shown in Appendix 6.4.

The model presents a proof of concept for a DBN based decision support tool for regulators and proponents, as well as a tool for 'what-if' scenario analysis, evaluating whole-of-system effects and integrating data and knowledge for scientists. This project is a collaboration between AIMS, QUT, ECU and UWA.

4.2 Conclusions

Dredging activities may have drastic impacts on marine organisms, particularly the benthos. Environmental Windows (EWs), or the cessation of dredging during ecologically sensitive periods can be an effective management tool if they are set properly. In addition to an understanding of environmental conditions, this requires location-specific knowledge of the timing of sensitive periods in the life histories of the organisms present.

The selection of effective EWs is highly dependent on the particular habitat and species present. These may be highly diverse, with correspondingly diverse life history characteristics and variable vulnerabilities to disturbance. Thus, the first step in the selection of EWs for dredging is to assess the ecological, social and economic 'value' of the species present in order to prioritise protection. Finally, the vulnerability of these species is assessed based on their life history characteristics and sensitivity to environmental change.

Marine invertebrates can play important roles in the habitats in which they occur. The filter feeders, in particular, are a highly diverse and ecologically important group, providing food and shelter for other sessile and mobile organisms. These can also be of great economic importance. For example, sponges have been used for the production of chemicals for biomedical research (Fromont et al. 2006). As such, in habitats such as temperate reefs which are dominated by sponges and other filter feeders, these should be considered when making protection and management decisions.

Habitat forming taxa such as seagrasses and macroalgae should also take priority for protection and management. Seagrass meadows are highly important habitats in shallow coastal and estuarine ecosystems (Kemp 1983). They provide food, shelter and other ecological services to many ecologically and commercially important marine organisms (Costanza 1997, Beck et al. 2001) and are amongst the most productive aquatic communities (Larkum & West 1983). Similarly, macroalgal beds are extremely ecologically important in most shallow temperate marine ecosystems, supporting diverse communities of fish and invertebrates (Fletcher 1987, Wernberg et al. 2003, Irving et al. 2004, Tuya et al. 2008).

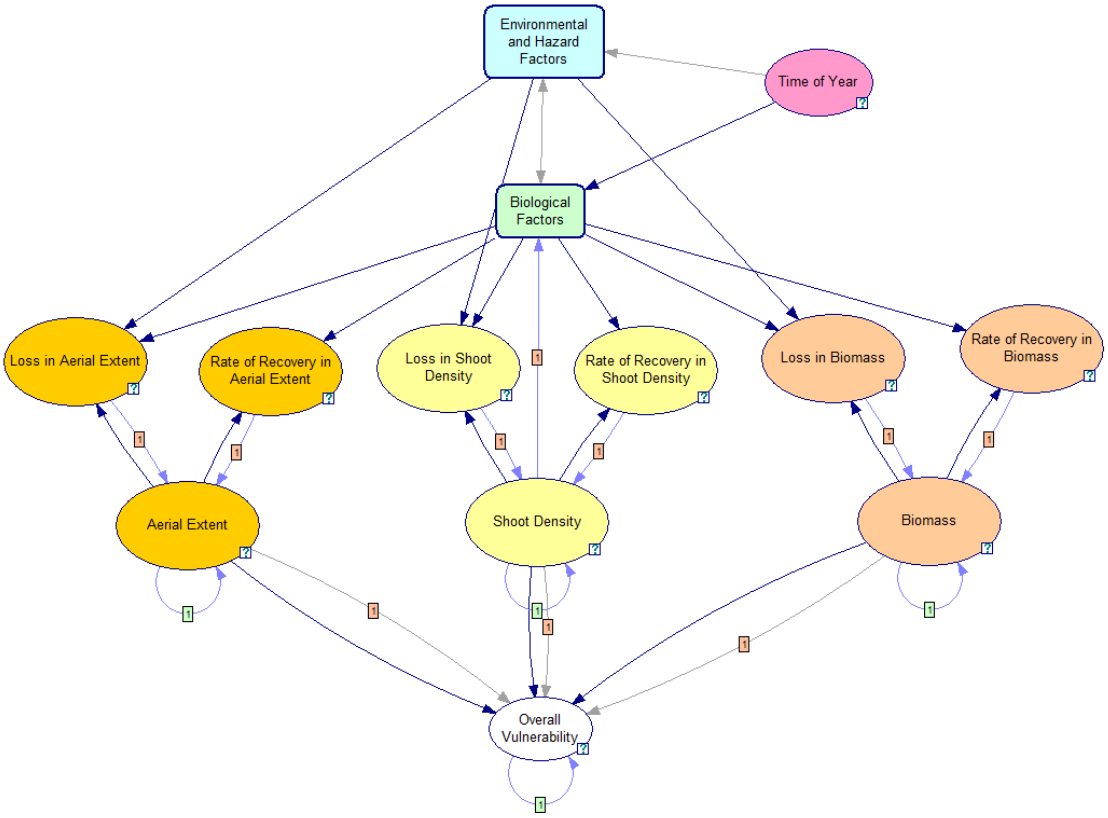

Figure 3. Upper level structure of the seagrass vulnerability model. The ovals represent different nodes or factors in the model, and the arrows causal influence. The notation '1' on an arrow denotes a temporal relationship between the current time step at a node and the next time step at a connected node, enabling the model to capture temporal variations in vulnerability.

In Western Australia, there is the opportunity to improve and refine the use of EWs by identifying and understanding how dredging may impact key ecological processes in nearshore marine ecosystems. In WA, it is known that many marine organisms exhibit an increased vulnerability to disturbance during the late spring to early autumn period (Oct–April) due to the timing of sensitive life history periods (periods of reproduction and recruitment), such that winter is a period of the year when dredging would pose the lowest risk to critical life cycle processes for a number of taxa. However this does not hold true for ephemeral seagrasses. Furthermore, local information on potentially critical periods and detailed knowledge of life history characteristics are missing for many dominant WA species of invertebrates, seagrasses and macroalgae. Thus, there is a need to improve our knowledge of sensitive periods (reproductive periods, in particular) in the life histories of many WA marine species by undertaking a series of basic biological studies. We have also explored Dynamic Bayesian Networks as an adaptive tool to assist in decision-making around EW's, investigating the effects of the timing and duration of dredging on ephemeral and persistent seagrasses in this pilot study. From the results obtained we recommend a full detailed study using DBN-type approaches be undertaken on a range of biota.

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6 Appendices

6.1 General vulnerability index for ecologically important invertebrate taxa based on life history characteristics.

Vulnerability Index	H = High, M = Medium, L = Low
Habitat or community type	FF = FilterFeeder community habitat, SCR = subtidal coral reef, RR= rocky reef
Reproductive mode	VEG= Vegetative, SEXU = Sexual
Larval Feeding mode/connectivity	LEC = Lecitotrophs, BCS = Broadcast spawners, BRO = Brooders, PLANK = Planktotrophs
Adult Movement	SES = Sessile. MOB = Mobile
Adult feeding mode	FF = Filter feeding , DF = Deposit feeding, AUTO = Autrotroph, GRA = Grazing , PRED = Predator
Phylum	Ascidians = ASC, Bivalves = BIV, Bryozoans = BRY, CNID = cnidaarians, Crustaceans = CRUS, Echinoderms = ECH, Molluscs = MOLL, Sponges = SPON,

VI	Phylum	Class/ Growth form	Habitat or community	Reproductive mode	Adult feeding mode	Larval feeding mode connectivity	Adult movement	Juvenile considerations	Longevity	Times to dredge
H	SPON	ENCRUST	FF, SCR, RR	SEXU VEG	FF small particles	LEC	SES	Vulnerable when young/access to space/ smothering	Annual to very long lived - indeterminate	Not during larval settlement
H	SPON	ENCRUST	FF, SCR, RR	SEXU VEG	FF (small particles) AUTO	LEC	SES	Vulnerable when young/access to space/ smothering	Annual to very long lived - indeterminate	Not during larval settlement
M	SPON	Erect	FF, SCR, RR	SEXU VEG	FF (small particles), AUTO	LEC	SES	Vulnerable when young/access to space/ smothering	Annual to very long lived- indeterminate	Not during larval settlement
M/H	SPON	Cup	FF, SCR, RR	SEXU VEG	FF (small particles), AUTO	LEC	SES	Vulnerable when young/access to space/ smothering	Annual to very long lived- indeterminate	Not during larval settlement
M	ASC	Solitary	FF, SCR, RR	SEXU	FF (large particles),	LEC BCS	SES	Vulnerable when young/access to space/ smothering	Years – decades	Not during larval settlement
H	ASC	COLONIAL	FF, SCR, RR	SEXU VEG	FF (large particles) AUTO	Mostly BR	SES	Vulnerable when young/access to space/ smothering	Years – indeterminate/ some annual	Not during larval settlement
H	BRY	Colonial (includes encrusting and erect forms)	FF, SCR, RR	SEXU VEG	FF (medium particles)	BR	SES	Vulnerable when young/access to space/ smothering	Months to few years	Not during larval settlement

H	MOLL	Gastropods	FF, SCR, RR, SS	SEXU	GRA, PRED, DF	BR, egg layers, L	Weakly MOB	Vulnerable when young/access to space/ smothering also smothering of egg mass	Variable, years – decades	Not during larval settlement or egg mass deposition
H	MOLL	Gastropods	FF, SCR, RR, SS	SEXU	GRA, PRED, deposit feeders	BR, Egg layers, PLANK	Weakly MOB	Vulnerable when young/access to space/ smothering also smothering of egg mass	Variable, years – decades	Not during larval settlement or egg mass deposition
M/H	MOLL	Gastropods	FF, SCR, RR, SS	SEXU	GRA, PRED, deposit feeders	BCS	Mostly weakly MOB	Vulnerable when young/access to space/ smothering	Variable, years to decades	Not during larval settlement
M/L	MOLL	Bivalves	FF, SCR, RR, SS	SEXU	FF, DF	BR	Mostly SES (except e.g. scallops)	Vulnerable when young/access to space/ smothering	Variable, years to decades	Not during reproduction/egg laying
M/L	MOLL	Bivalves	FF, SCR, RR, SS	SEXU	FF, DF, partially AUTO	BCS PLANK	Mostly SES	Vulnerable when young/access to space/ smothering	Variable, years to decades	Not during larval settlement
M/L	MOLL	Bivalves	FF, SCR, RR, SS	SEXU	FF, DF	BCS L	Mostly SES	Vulnerable when young/access to space/ smothering	Variable, years to decades	Not during larval settlement
L except for egg masses	MOLL	Cephalopods	Pelagic, RR	SEXU	PRED	Egg layers	Highly MOB, some pelagic	Egg masses vulnerable to smothering	Annual	Not during reproduction/egg laying
M	MOLL	Chitons	RR, SCR	SEXU	GRA	BCS L	Limited adult mobility	Vulnerable to smothering as juveniles	Years to decades	Not during larval settlement

M	ECH	Asteroids	SCR, RR, SS	SEXU,VEG	PRED, DF	BCS LEC	Weakly MOB	Vulnerable when young	Years to decades, asexual forms indeterminate	Avoid reproductive periods
M/L	ECH	Asteroids	SCR, RR, SS	SEXU,VEG	PRED, DF	BCS, PLANK	Weakly MOB	Vulnerable when young	Years to decades, asexual forms indeterminate	Avoid reproductive periods
M/H	ECH	Asteroids	SCR, RR, SS	SEXU,VEG	PRED, DF	BR, LEC	Weakly MOB	Vulnerable when young	Asexual forms indeterminate	Avoid reproductive periods
M	ECH	Ophiuroids	FF, SCR, RR, SS	SEXU,VEG	FF, DF	BCS, LEC	Weakly MOB	Vulnerable to smothering, sediment clogging	Years, asexual forms indeterminate	Avoid reproductive periods
M/ L	ECH	Ophiuroids	FF, SCR, RR, SS	SEXU,VEG	FF, DF	BCS, PLANK	Weakly MOB	Vulnerable to smothering, sediment clogging	Years, asexual forms indeterminate	Avoid reproductive periods
M/ H	ECH	Ophiuroids	FF, SCR, RR, SS	SEXU	FF, DF	BR, LEC	Weakly MOB	Vulnerable to smothering, sediment clogging	Years	Avoid reproductive periods
M/ H	ECH	Echinoids	RR, SCR	SEXU	GRA, HERB	BCS, LEC	Weakly MOB	Vulnerable to smothering	Years to decades	Avoid reproductive periods
M/ L	ECH	Echinoids	RR, SCR	SEXU	GRA, HERB	BCS, PLANK	Weakly MOB	Vulnerable to smothering	Years to decades	Avoid reproductive periods
M	ECH	Irregular Echinoids	SS	SEXU	DF	BCS, PLANK	Weakly MOB	Vulnerable to smothering	Years to decades	Avoid reproductive periods

M/L	ECH	Holothuroids	SS, RR, SCR	SEXU	FF, DF	BCS, LEC	Weakly MOB/SES	Vulnerable to smothering, sediment clogging	Years to decades, asexual indeterminate	Avoid reproductive periods
M/L	ECH	Holothuroids	SS, RR, SCR	SEXU	FF, DF	BCS, PLANK	Weakly MOB/SES	Vulnerable to smothering, sediment clogging	Years to decades, asexual indeterminate	Avoid reproductive periods
M/L	ECH	Holothuroids	SS, RR, SCR	VEG	DF	BR	Weakly MOB/SES	Vulnerable to smothering	Indeterminate	Avoid reproductive periods
M/H	ECH	Crinoids	SS, RR, SCR	SEXU	FF	BCS, LEC	Functionally SES	Vulnerable to smothering, sediment clogging	Years to decades	Avoid reproductive periods
H	CRUST	Barnacles	RR, SCR	SEXU	FF	BCS, PLANK	SES	Vulnerable to smothering, sediment clogging	Unknown	Avoid reproductive periods
M/L	CRUST	Decapods	RR, SCR,SS	SEXU	PRED, GRA	BCS, BR, PLANK	MOB	Vulnerable to smothering, sediment clogging	Years to decades	Avoid reproductive periods
M/H	Non-coral CNID		RR, SCR,SS	SEXU,VEG	PRED, FF some partially AUTO	BCS, BR, LEC	SES	Vulnerable to smothering, sediment clogging	Years to decades	Not during larval settlement
M/H	Pelagic CNID	Sea jellies	Pelagic	SEXU,VEG	PRED	BCS, PLANK	Pelagic, limited mobility	Benthic juveniles vulnerable to smothering, adults to sediment clogging	Weeks to months	Avoid reproductive periods

6.2 Vulnerability index based on LHCs and timing of reproduction for WA representative invertebrate species or for the same species elsewhere. Sub-groupings are indicated when vulnerability indices vary within a phylum.

Vulnerability Index	Phylum	Representative taxa based on commonness, functional, habitat defining characteristics	Timing of reproduction	Justification for representative species or taxa
Medium/High	Sponges	*Pione velans, Mycale* spp., *Chondrilla australiensis, Lamellodysidea herbacea, Clinona spp.;* At least partially autotrophic: *Pericharax heteroraphis, Neopetrosia exigua, Carteriospongia flabellifera, Xestospongia testudinaria;* Heterotrophic: *Clathria (Thalysias) reinwardti, Axos flabelliformis, Lanthella basta, Stylissa flabelliformis, Xenospongia patelliformi, Haliclona* spp.	Reproduce annually; *Pione velans* oocyte development in May (Fromont et al. 2005); Reproduce annually late summer and autumn coincident with moon/tide phase (Usher et al. 2004); *Xestospongia testudinaria,* Orpheus Island, GBR, spawns in Oct.- Nov. when temperatures are warm, with possible influence on exact timing by lunar cycle and tides (Fromont & Bergquist 1994).	Advised by Jane Fromont (WAM) and Fiona Webster (AIMS); Well studied, common WA sponges (Usher et al. 2004).
Medium/High	Ascidians	*Pyura, Polycarpa* spp., *Phallusia, Cnemidocarpa, Distaplia stylifera, Nepthesis fasacicularis, Aplidium, Didemnum, Pseudodistoma*	No local information.	Commonly recorded species from Damper Peninsula (Keesing et al. 2011).
High	Bryozoans	*Adeona grisea, Bugula, Celloporaria, Iodictyum, Triphyllozoon*	*Bugula neritina*, Williamstown, VIC: reproduce in Jan-Feb (Marshall et al. 2003).	Well known Bryozoa genera (Andy Davis pers. Comm.) including from WA (Edgar 1997).
High	Molluscs (Cowries)	*Notocypraea, Zoila* and *Austrocypraea*	*Notocypraea piperita* and *Austrocypraea reevei,* WA: females on eggs observed Sept. - Jan., egg laying presumed to being in Aug. *Zoila venusta* egg masses observed late Oct. - Feb. (Wilson 1985).	Iconic endemic Western Australian cowries (Wilson 1985).
Medium/High	Molluscs (Gastropods)	*Nerita albicilla, Nerita antiquata, Chicoreus cervicornis, Murex acanthostephe, Tectus pyramis, Trochus histrio, Astralium rotularium, Turbo bruneus*	*Nerita albicill,* Hong Kong: spawn in Sept. (Yeung 2006).	Common gastropods found in Pilbara and Kimberley (Keesing et al. 2011).
Medium/Low	Molluscs (Bivalves)	*Austrocardiella, Condylocardia, Cunanax, Ephippodontoana, Mysella;*	*Saccostrea:* (Talbot 1986).	Peter Middelfart, Australian Museum; *Saccostrea* are a

		Oysters*: Saccostrea,* Clams*: Tridacna spp.; Lecithotrophs: Ennucula cummingii, E. superba*		common oyster in NW WA; *Tridacna* spp. are partially or predominantly autotropic.
Low except for egg masses (opportunity for EW)	Molluscs (Cephalopods)	*Euprymna tasmanica, Sepia apama, Octopus tetricus, O. maorum, Sepioteuthis australis*	*Euprymna tasmanica* is a multiple spawner (Steer et al. 2004); *Octopus maorum*, TAS: spawn during austral winter (Grubert & Wadley 2000); *Sepioteuthis australis,* Tas, aggregate in shallow, protected waters for spring-summer spawning period (Steer et al. 2002); *Sepia apama,* Spencer Gulf, SA: aggregate during winter to spawn (Hall & Hanlon 2002).	These are the most common species of cephalopods in Western Australia and cover all three main types.
Medium	Molluscs (Chitons)	*Acanthopleura gemmata, Lucilina fortilirata, Ischnochiton virgatus, Clavarizona hirtosa, Onithochiton occidentalis*	*Acanthopleura gemmata*, GBR: continuous gamete development and release over 6-month spawning season from early summer-late autumn (Barbosa et al. 2009).	Common chitons found in Temperate and tropical WA (Wells & Sellers 1987).
Medium/Low	Echinoderms (Asteroids: broadcast spawners/ lecithotrophs/ planktotrophs)	*Fromia, Nardoa, Gomophia, Linckia laevigata, Pentaceraster, Protoreaster*	Observations of *Fromia* spawn coinciding with mass coral spawn at Abrolhos Is. (Marsh 1994); *Linckia laevigata,* Heron Is., GBR: spawning observed in October (Laxton 1974); Asan, Guam: spawn during summer (Yamaguchi 1977).	Common sea star genera in northwest WA with lecithotrophic *(Fromia, Nardoa, Gomophia)* or planktotrophic *(Linckia laevigata, Pentaceraster, Protoreaster)* larval development.
Medium/High	Echinoderms (Asteroids: brooders/ lecithotrophs)	Euretaster insignis	No information.	Common sea star genera in NW WA with brooding direct larval development (Keesing unpublished).
Medium/Low	Echinoderms (Ophiuroids: broadcast spawners/ lecithotrophs/ planktotrophs)	*Ophionereis dubia, O. semoni, Ophiactis maculosa, O. savignyi*	*Ophionereis schayeri* in NSW have a major spawning period in summer (Jan-Feb; (Selvakumaraswamy & Byrne 1995); *Ophiactis resiliens,* Sydney, NSW: annual spawning period from May through Sept., with peak gamete release from May- July (Falkner & Byrne 2003).	These species occur in the Kimberley (Keesing et al. 2011).
Medium/High	Echinoderms (Echinoids: broadcast spawners/ lecithotrophs)	*Heliocidaris erythrogramma*	Spawning occurs in December–March over a large part of the species' eastern Australian distribution, including Sydney (Williams & Anderson 1975, Laegdsgaard et al. 1991), Hobart (Dix 1977) and Melbourne (Constable 1989, Keesing 2001).	Common WA species (Keesing 2006).

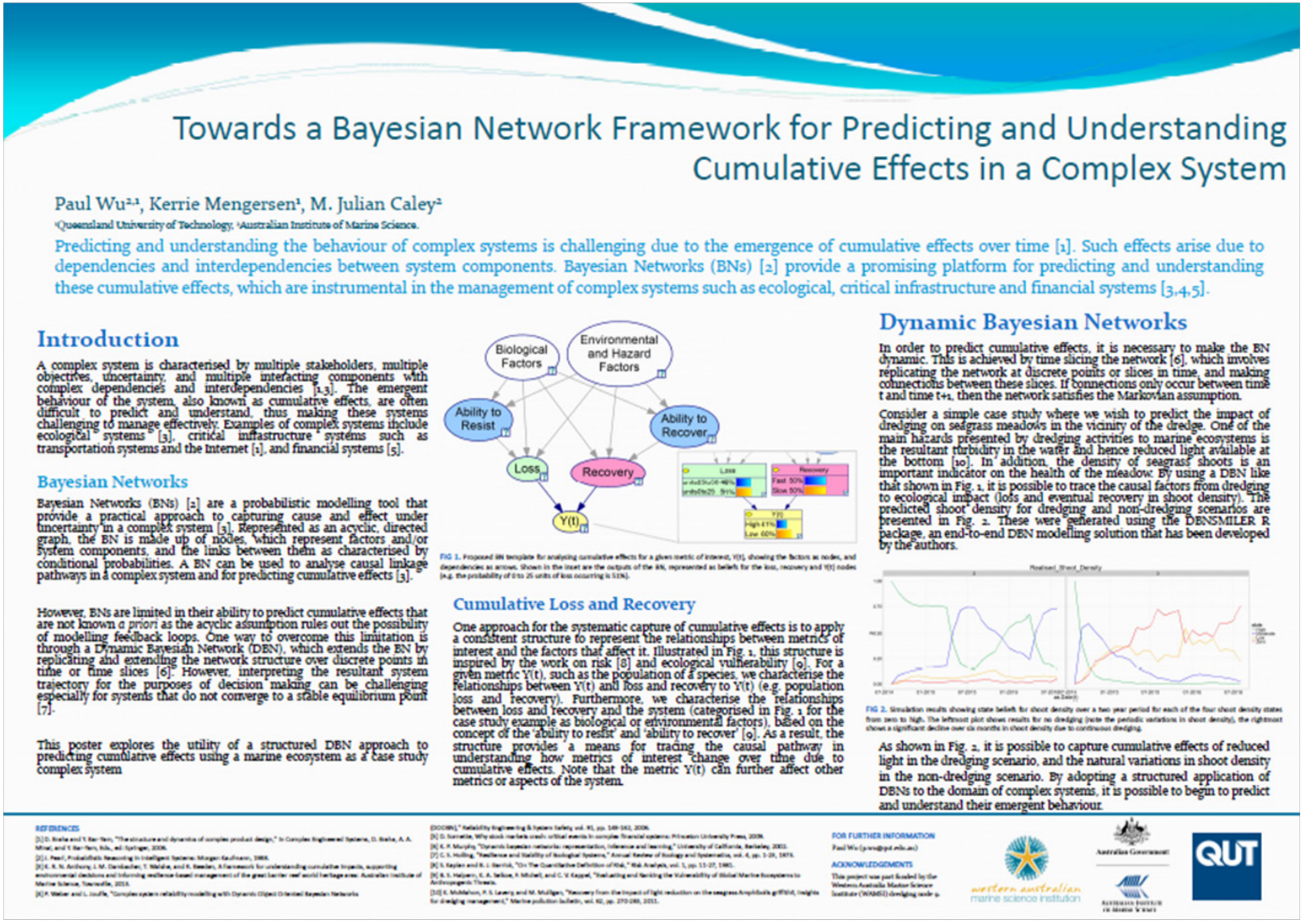
Medium/Low	Echinoderms (Echinoids + Irregular Echinoids: broadcast spawners/ planktotrophs)	Echinoids: *Diadema savignyi, Echinostrephus molaris, Echinometra mathaei, Centrostephanus tenuispinus;* Irregular Echinoids*: Breynia desori, Lovenia elongata, Clypeaster virescens*	*Diadema savignyi,* South Africa: spawn during summer (Dec–March/April; (Drummond 1995)). *Echinometra mathaei*, Rottnest Is., WA: continuous spawning (Pearse & Phillips 1968).	Common WA species.
Medium/Low	Echinoderms (Holothuroids)	*Psolus* spp.*, Psolidium* sp.*, Holothuria leucospilota, Holothuria atra, Stichopus chloronatus*	*Holothuria leucospilota:* Summer months - Nov to April (Cook Islands) (Drumm and Loneragan 2005); Holothuria atra: June/July, Jan/Feb (GBR) (Harriott 1985)., May to Sept (Taiwan) (Chao et al. 1994); most Fission in winter (Uthicke); *Stichopus chloronatus:* April spawning and fission (Darwin)(Purwati 2004), winter fission (GBR), Summer spawning (GBR) (Ulthicke).	Species well represented in WA (Mackenzie & Whitfield 2011).
Medium/High	Echinoderms (Crinoids)	*Comanthus, Clarkcomanthus, Comaster, Comatella, Comatula, Heterometra, Zygometra*	No information.	Common genera from WA (Keesing et al. 2011).
High	Crustaceans	*Striatobalanus amaryllis, Amphibalanus poecilotheca, Acasta*	No information.	Common genera from WA (et al. 2011).
Medium/Low	Crustaceans	*Portunus rubromarginatus, P. pelagicus, Metapenaeopsis toloensis, Scylla serrata, S. olivacea, Penaeus semisulcatus*	*Penaeus semisulcatus:* major spawning peak in Aug.- Nov., minor peak in Jan–March (Gulf of Carpentaria, Northwestern Gulf, Aus; (Crocos 1987, Crocos & Van der Velde 1995)). *Scylla serrata:* spawning peak in Oct–Nov in northern Australia, migrate offshore to spawn (Hill 1994); *Portunus pelagicus:* Spawn in Oct–Jan (Koombana Bay and Cockburn Sound, WA; (Potter & De Lestang 2000)).	These are the most common decapods among three recent WA studies (Keesing et al. 2011).
Medium/High	Non-coral and pelagic Cnidarians	Gorgonian*: Subergorgia suberosa, Alertigorgia* spp. including *A. orientalis;* Soft corals *Chromonephthea* spp. incl. *C. curvata, C. fruticosa, C. ostrina, Lobophytum crassum, Sinularia* spp.; Sea fan: *Ctenocella pectinata;* Sea whip: *Junceela fragilis; Sea jelly: Crambione mastigophora.*	*Lobophytum compactum* spawns annually, on the 4th day after the November full moon at Orpheus Island, GBR (Alino & Coll 1989).	*S. suberosa* was common on the Dampier Peninsula (Keesing et al. 2011) and the effects of sedimentation on it have been studied (Tseng et al. 2011); *Crambione mastigophora* is abundant on the coast in NW Australia (Marsh & Slack-Smith 2010).

6.3 Summary of seagrass response to light limitation in Western Australia.

Seagrass Species	Study location, type and duration	Light levels (% SI) and turbidity characteristics	Main findings: Effect of conditions on seagrasses and their recovery after the study	Reference
Amphibolis griffithii	Jurien Bay, WA Shading of Seagrass for 3, 6, 9 months under two irradiance levels. Leaf biomass measured.	13-19% SI 5-11% SI SI designed to mimic wide scale dredging activity.	3 months shading, caused (max) 72% reduction in leaf biomass. Full recovery after 300 days. Shading of 6 and 9 months past recovery threshold: no recovery of plants after 730 days.	(McMahon et al. 2011)
Halophila ovalis	Swan River, WA *In situ* surveys plus tank studies with artificial light, temperature (25˚C) and salinity (35). Experiments run for 6 weeks.	Minimum light requirement for continued growth was measured at 40 µmol m^{-2} s^{-1} or 2% of full sunlight PAR. 99% of distribution in the field was in water less than 2m due to light limitation.	Main growth in summer with low turbidity, high temperature and marine salinity; winter had lower growth. High flow event in 1981 resulting in increased turbidity caused severe declines in biomass. Light determined to be the critical limiting parameter for distribution and growth.	(Hillman et al. 1995)
Halophila ovalis	Qld, Aus *In situ* shading of seagrass. Seagrass monitored over 30 day period.	The quantity of light was extremely variable, with mean daily irradiances between 9–12 mol photons m^{-2} day^{-1}, and an overall range of 0.05–42 mol photons m^{-2} day^{-1}	Duration of survival under zero light conditions was 15 days. Plants recovered when 7% SI was restored. 30 days complete mortality in conditions above threshold. Change in leaf physiology (e.g. amino acid content, chlorophyll content and $\delta^{13}C$) occurred before morphological changes (e.g. biomass, shoot density, canopy height) or die-off, and were thus considered to be potential indicators of impending seagrass die-off during light deprivation.	(Longstaff & Dennison 1999)
Posidonia australis	Jervis bay, NSW, Aus Temperate. Shoot Density measured after 90 day period of sub-lethal light exposure. Recovery measured over 510 day period. 3-5m depth.	Threshold = 9.2 %SI Light reduced to 10% SI for the duration of the experiment.	Reductions in shoot density and leaf productivity. No recovery in shoot density after 510 days of normal conditions. Seasonal difference in the effect of shading: greater impact during early summer than during late summer (rhizome reserves).	(Fitzpatrick & Kirkman 1995)

Seagrass Species	Study location, type and duration	Light levels (% SI) and turbidity characteristics	Main findings: Effect of conditions on seagrasses and their recovery after the study	Reference
Posidonia sinuosa	Cockburn Sound, WA *In situ* shading at depths of 4 and 8 m. Three levels of shading (low (LS), medium (MS), high (HS) and control). Shoot density and biomass measured after 198 days of shading. Recovery measured after 400 days	4m : Control : 29%SI LS : 24%SI MS : 7% SI HS : 2% SI 8m Control : 14%SI LS : 12-14%SI MS: 2-4% SI	Shoot density was strongly negatively correlated to light reaching seagrass. Leaf length and growth also decreased with shading time. Recovery minimal: predicted 3.5-5 years.	(Collier et al. 2009)
Posidonia sinuosa	Albany, WA *In situ* shading with shade cloth for 148 days. Recovery measured over 245 days.	10-1%SI	Shading reduced shoot density, primary production and leaf production per shoot. Shoot density and productivity of P. sinuosa shaded for 307 days and 393 days were 10% and 8%, respectively, of unshaded controls. After 245 days shoot density and leaf productivity did not return to control levels.	(Gordon et al. 1994)

6.4 Detailed description of how Bayesian Networks are to model complex systems.



Towards a Bayesian Network Framework for Predicting and Understanding Cumulative Effects in a Complex System

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Predicting and understanding the behaviour of complex systems is challenging due to the emergence of cumulative effects over time [1]. Such effects arise due to dependencies and interdependencies between system components. Bayesian Networks (BNs) [2] provide a promising platform for predicting and understanding these cumulative effects, which are instrumental in the management of complex systems such as ecological, critical infrastructure and financial systems [3,4,5].

Introduction

A complex system is characterised by multiple stakeholders, multiple objectives, uncertainty, and multiple interacting components with complex dependencies and interdependencies [1,3]. The emergent behaviour of the system, also known as cumulative effects, are often difficult to predict and understand, thus making these systems challenging to manage effectively. Examples of complex systems include ecological systems [3], critical infrastructure systems such as transportation systems and the Internet [1], and financial systems [5].

Bayesian Networks

Bayesian Networks (BNs) [2] are a probabilistic modelling tool that provide a practical approach to capturing cause and effect under uncertainty in a complex system [3]. Represented as an acyclic, directed graph, the BN is made up of nodes, which represent factors and/or system components, and the links between them as characterised by conditional probabilities. A BN can be used to analyse causal linkage pathways in a complex system and for predicting cumulative effects [3].

However, BNs are limited in their ability to predict cumulative effects that are not known *a priori* as the acyclic assumption rules out the possibility of modelling feedback loops. One way to overcome this limitation is through a Dynamic Bayesian Network (DBN), which extends the BN by replicating and extending the network structure over discrete points in time or time slices [6]. However, interpreting the resultant system trajectory for the purposes of decision making can be challenging especially for systems that do not converge to a stable equilibrium point [7].

This poster explores the utility of a structured DBN approach to predicting cumulative effects using a marine ecosystem as a case study complex system

FIG 1. Proposed BN template for analysing cumulative effects for a given metric of interest, Y(t), showing the factors as nodes, and dependencies as arrows. Shown in the inset are the outputs of the BN, represented as beliefs for the loss, recovery and Y(t) nodes (e.g. the probability of 0 to 25 units of loss occurring is 51%).

Cumulative Loss and Recovery

One approach for the systematic capture of cumulative effects is to apply a consistent structure to represent the relationships between metrics of interest and the factors that affect it. Illustrated in Fig. 1, this structure is inspired by the work on risk [8] and ecological vulnerability [9]. For a given metric Y(t), such as the population of a species, we characterise the relationships between Y(t) and loss and recovery to Y(t) (e.g. population loss and recovery). Furthermore, we characterise the relationships between loss and recovery and the system (categorised in Fig. 1 for the case study example as biological or environmental factors), based on the concept of the 'ability to resist' and 'ability to recover' [9]. As a result, the structure provides a means for tracing the causal pathway in understanding how metrics of interest change over time due to cumulative effects. Note that the metric Y(t) can further affect other metrics or aspects of the system.

Dynamic Bayesian Networks

In order to predict cumulative effects, it is necessary to make the BN dynamic. This is achieved by time slicing the network [6], which involves replicating the network at discrete points or slices in time, and making connections between these slices. If connections only occur between time t and time t+1, then the network satisfies the Markovian assumption.

Consider a simple case study where we wish to predict the impact of dredging on seagrass meadows in the vicinity of the dredge. One of the main hazards presented by dredging activities to marine ecosystems is the resultant turbidity in the water and hence reduced light available at the bottom [10]. In addition, the density of seagrass shoots is an important indicator on the health of the meadow. By using a DBN like that shown in Fig. 1, it is possible to trace the causal factors from dredging to ecological impact (loss and eventual recovery in shoot density). The predicted shoot density for dredging and non-dredging scenarios are presented in Fig. 2. These were generated using the DBNSMILER R package, an end-to-end DBN modelling solution that has been developed by the authors.

FIG 2. Simulation results showing state beliefs for shoot density over a two year period for each of the four shoot density states from zero to high. The leftmost plot shows results for no dredging (note the periodic variations in shoot density), the rightmost shows a significant decline over six months in shoot density due to continuous dredging.

As shown in Fig. 2, it is possible to capture cumulative effects of reduced light in the dredging scenario, and the natural variations in shoot density in the non-dredging scenario. By adopting a structured application of DBNs to the domain of complex systems, it is possible to begin to predict and understand their emergent behaviour.

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FOR FURTHER INFORMATION

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6.5 Western Australian seagrass life history characteristics and vulnerability to dredging.

Vulnerability Index	Genus	Adult size	Growth rate	Time to sexual maturity	Seed bank	Seasonal considerations	Longevity	Times to dredge	References
Medium	*Posidonia*	Large	Moderate	Years	Absent	Seasonal variation in flowering, seed release and recruitment	Long turnover times (years)	Avoid reproduction and recruitment	(Kilminster et al. 2015)
Medium/low	*Amphibolis*	Medium	Moderate	Years	Absent	Seasonal variation in flowering, seedling release and recruitment	Long turnover times	Avoid reproduction and recruitment	(Kilminster et al. 2015)
High	*Thallasia*	Medium	Slow	Years	Absent	Continuous reproduction, seasonal growth patterns	Long turnover times (years)	Avoid reproduction and recruitment	(Kilminster et al. 2015)
Medium/low	*Enhalus*	Large	Rapid	Years	Absent	Continuous reproduction, seasonal growth patterns	Long turnover times (years)	Avoid reproduction and recruitment	(Kilminster et al. 2015)
Medium/low	*Halophila*	Small	Rapid	Weeks - months	Present	Seasonal variation in flowering, seed release and recruitment	Short turnover times (months)	Avoid reproduction and recruitment	(Kilminster et al. 2015)
Medium/low	*Zostera*	Small	Rapid	Months-years	Present	Seasonal variation in flowering, seed release and recruitment	Medium turnover times (months – years)	Avoid reproduction and recruitment	(Kendrick et al. 2012)

6.6 Macroalgal life history characteristics and vulnerability to dredging.

Vulnerability Index	Phylum	Sub-group	Reproductive mode	Adult feeding mode	Larval feeding mode/ Connectivity	Adult movement	Seasonal considerations	Longevity	Times to dredge	References
Low	Phaeophyta	*Sargassum*	Vegetative from stumps, Sexual	Autotrophs	Autotrophs, limited mobility	Sessile, clonal	Seasonal variation in growth, biomass and release of zygotes.	Adult: Average: 1-2 years Max.: 8 years	Avoid settlement, recruitment and regeneration.	(Kendrick & Walker 1994, 1995)
Low		*Ecklonia*	Sexual	Autotrophs	Autotrophs, flagellated (motile over small distances)	Sessile	Seasonal variation in growth, biomass and release of zoospores.	Adult: Average: 1-2 years Max.: 12 years	Avoid settlement, recruitment and regeneration.	(Wernberg & Goldberg 2008, Mohring et al. 2013a, Mohring et al. 2013b)
Low	Chlorophyta	*Halimeda*	Vegetative, Sexual	Autotrophs	Autotrophs, carpospores and male sperm mobile in water column.	Sessile, but can regrow from remaining thallus.	Timing of reproduction unknown, but could be year round.	Frond longevity: 3-5 months	Most sensitive when spores settling and recruiting.	(Hillis-Colinvaux 1986)
Low	Rhodophyta	Crustose coralline algae	Vegetative, Sexual	Autotrophs	Autotrophs	Sessile	Timing of reproduction unknown, but could be year round.	Adult: years	Most sensitive when spores settling and recruiting,	(Adey & Macintyre 1973, Adey & Vassar 1975)