

1 Effects of dredging on critical ecological processes for marine invertebrates, seagrasses and  
 2 macroalgae, and the potential for management with environmental windows using Western  
 3 Australia as a case study

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6

## 7 Abstract

8 Dredging can have significant impacts on benthic marine organisms through mechanisms  
 9 such as sedimentation and reduction in light availability as a result of increased suspension of  
 10 sediments. Phototrophic marine organisms and those with limited mobility are particularly at  
 11 risk from the effects of dredging. The potential impacts of dredging on benthic species

12 depend on biological processes including feeding mechanism, mobility, life history  
13 characteristics (LHCs), stage of development and environmental conditions. Environmental  
14 windows (EWs) are a management technique in which dredging activities are permitted  
15 during specific periods throughout the year; avoiding periods of increased vulnerability for  
16 particular organisms in specific locations. In this review we identify these critical ecological  
17 processes for temperate and tropical marine benthic organisms; and examine if EWs could be  
18 used to mitigate dredging impacts using Western Australia (WA) as a case study. We  
19 examined LHCs for a range of marine taxa and identified, where possible, their vulnerability  
20 to dredging. Large gaps in knowledge exist for the timing of LHCs for major species of  
21 marine invertebrates, seagrasses and macroalgae, increasing uncertainty around their  
22 vulnerability to an increase in suspended sediments or light attenuation. We conclude that  
23 there is currently insufficient scientific basis to justify the adoption of generic EWs for  
24 dredging operations in WA for any group of organisms other than corals and possibly for  
25 temperate seagrasses. This is due to; 1) the temporal and spatial variation in the timing of  
26 known critical life history stages of different species; and 2) our current level of knowledge  
27 and understanding of the critical life history stages and characteristics for most taxa and for  
28 most areas being largely inadequate to justify any meaningful EW selection. As such, we  
29 suggest that EWs are only considered on a case-by-case basis to protect ecologically or  
30 economically important species for which sufficient location-specific information is  
31 available, with consideration of probable exposures associated with a given mode of  
32 dredging.

33

34 **Keywords:** Dredging; sedimentation; environmental windows; marine biota; invertebrates;  
35 seagrass, macroalgae

36

37 **Highlights:**

38 • Reducing dredging during sensitive life history periods may reduce dredging impacts.

39 • Selection of Environmental Windows relies on accurate species- and location-specific  
40 information.

41 • Knowledge gaps exist for life histories of marine organisms in Western Australia.

42 • Environmental Windows are best considered on a location-specific basis for  
43 important species.

## 44 **1. Introduction**

45 Dredging is the excavation and relocation of sediment from an area to improve navigational  
46 access, for land reclamation and to allow for the development of coastal infrastructure  
47 (PIANC, 2010). Dredging can impact marine ecosystems in numerous ways. Dredging can  
48 increase turbidity and sedimentation above natural background peak and duration levels,  
49 reducing light availability and potentially burying benthic communities (EPA, 2011). In  
50 addition, dredging can also impact the physical environment by altering bathymetry,  
51 potentially causing erosion under benthic communities (Erftemeijer et al., 2006). Dredging  
52 can also impact water quality by releasing contaminants or excess nutrients in sediments,  
53 particularly when conducted over contaminated sediments (Filho et al., 2004). The spatial  
54 and temporal scales of potential impacts also depend on durations and intensities of exposure,  
55 which vary depending on selection of dredge plant and local bathymetries, hydrodynamics  
56 and sediment properties, and areas are often classified around dredging activities based on  
57 estimated impact severity (e.g. areas of high impact/moderate impact/influence, EPA 2011).  
58 Dredging can therefore have adverse impacts on the marine environment, particularly on  
59 sessile benthic communities, if not managed effectively (Rogers, 1990; Desprez, 2000;  
60 Erftemeijer et al., 2012). In order to understand the vulnerability it is important to first  
61 understand their life history characteristics (LHCs) and identify sensitive life history stages  
62 (e.g. reproductive periods). We define vulnerability as “the extent to which a species  
63 experiences field effects of a stressor at the population level, as result of their species-specific  
64 ecological traits governing potential exposure to this stressor, toxicological sensitivity, and  
65 population recovery capacity” (sensu De Lange et al., 2010). Due to high spatial and  
66 temporal variability in the occurrence of ecologically critical periods, this knowledge is  
67 limited for many regions and many species.

68

69 Environmental windows (EWs) are a management strategy used to minimize the impacts of  
70 dredging on specific marine flora and fauna through temporal restrictions on intensive  
71 dredging activities, both at the sediment excavation site and at the sediment placement site,  
72 particularly if the latter is located in open water. EWs can be defined as periods during which  
73 dredging and the disposal of dredged material are expected to have fewer ecological impacts,  
74 whereas seasonal restrictions are periods when these activities should be limited or avoided  
75 (NRC, 2002). Setting effective EWs requires local ecological and environmental knowledge.  
76 A discrete period such as a mass spawning event for fish, corals or associated invertebrates is

77 an example of a predictable period during which a population may be particularly sensitive to  
78 dredging (Jones et al. 2015). The timing of such periods can be incorporated into the  
79 management of dredge operations to mitigate the effect on a particular species or group of  
80 species (Suedel et al., 2008). In some parts of the USA, several restrictions are imposed on  
81 dredging activities during spring and winter to protect certain species of fish (Reine et al.,  
82 1998; Suedel et al., 2008). For example, in San Francisco Bay, EWs are implemented to  
83 protect the commercially and ecologically important Pacific herring, that enter the bay in  
84 order to reproduce (Suedel et al., 2008). In Western Australia (WA), seasonal restrictions on  
85 dredging activities have been imposed to protect coral mass spawning events (Simpson,  
86 1985; 1991; Babcock et al., 1994; EPA, 2011). Monitoring of coral colonies to determine  
87 likely mass spawning events has been used to manage dredge operations in the north-west of  
88 WA (Styan and Rosser 2012).

89

90 Ecosystem-based management strategies are the most effective for managing environmental  
91 change (McLeod and Leslie, 2009). EWs can be applied in a broader context, addressing  
92 seasonal changes in the susceptibility of an ecosystem as a whole to dredging. For example,  
93 the tropical wet-dry climate of northern Australia produces discrete periods of higher  
94 turbidity in coastal waters during the wet season (November - April). During such intervals,  
95 marine organisms have adaptive strategies for coping with natural reductions in light levels  
96 and increases in turbidity (Lanyon and Marsh, 1995, Richards et al., 2015). As such, dredging  
97 operations may be best carried out during the wet season in this region, when turbidity levels  
98 are naturally higher, and restricted at the onset of the dry season, thereby avoiding an increase  
99 in turbidity levels outside of the natural range (van Senden et al., 2013). Furthermore,  
100 management strategies that incorporate the known tolerance of a species to impacts of  
101 dredging (e.g. reduced light) may allow for the application of environmental thresholds such  
102 that dredging activities can continue at particular times of year with little environmental  
103 impact. In Gladstone, Queensland, Australia, a light-based threshold using a rolling average  
104 was applied to protect seagrasses from the potential effects of sediment-related reductions in  
105 light levels during the growth season (July - December) (Chartrand et al., 2016). Combining a  
106 temporal restriction with a threshold approach is likely to be a lesser impost than a year-  
107 round threshold approach or a complete restriction on any dredging activities during the  
108 growth season.

109

110 The implementation of EWs has been difficult as the procedure for setting EWs has not  
111 followed a particular structure and has, at times, lacked scientific basis (NRC, 2002). EWs  
112 for dredging were initially established in the USA to protect periods of migration for  
113 commercially important taxa (shellfish larvae, finfish, etc.), however, the policy behind this  
114 strategy was disorganised and inconsistent, resulting in inflated dredging costs (Suedel et al.,  
115 2008). Some management measures require the cessation of dredging operations for short  
116 windows of time that, although effective for one life history process, may be too short for  
117 associated vulnerable early life history stages (Chevron 2009). Furthermore, the costs of  
118 stopping dredging can be substantial (potentially in the order of one to two million dollars per  
119 day per dredging vessel), especially if involving the seasonal demobilisation and  
120 remobilisation of an entire fleet to avoid a particular season (Suedel et al., 2008).

121

122 In this review, we have investigated the use of EWs from a critical life history phase  
123 perspective by attempting to identify periods of vulnerability for several groups of marine  
124 benthic organisms. We consider a myriad of life history stages and characteristics together on  
125 individual, multigenerational and population levels. Our first aim was to identify critical  
126 ecological processes for tropical and temperate habitats dominated by marine benthic biota  
127 other than fishes (i.e. invertebrates, seagrasses and macroalgae). Using Western Australia as a  
128 case study, our second aim was to determine the timing of these processes in order to identify  
129 the potential for EWs for dredging in this region. We then propose a general framework for  
130 the consideration of EWs for dredging.

131

## 132 **2. Methodology**

### 133 *2.1. Knowledge of life history characteristics*

134 In November 2013 and September 2014, workshops were held at the Commonwealth  
135 Scientific and Industrial Research Organisation (CSIRO), Floreat, Western Australia. These  
136 workshops brought together national and international marine scientists with expertise in the  
137 fields of marine ecology, botany, zoology and dredging. The workshop conducted an in-depth  
138 expert-based review and this was followed up by going back to workshop participants and  
139 other experts using a Delphi like approach (Linstone and Turoff 1975) to verify the accuracy  
140 and comprehensiveness of the life history characteristics (LHCs) and vulnerabilities to  
141 dredging associated with each LHC which were identified at the workshop. Based on the

142 collective expertise of the workshop attendees and other experts canvassed, as well as an  
143 expert advised literature searches in Google Scholar and ISI Web of Science using keywords  
144 relevant to the review (e.g. dredging, marine, invertebrates, seagrass, macroalgae plus  
145 combinations), we established a comprehensive assessment of LHCs and their associated  
146 vulnerabilities. In addition to this we conducted an extensive literature review and identified  
147 ecologically important benthic taxa from Western Australia specifically, and what was  
148 known of their LHCs. From this we identified the most favorable EWs for dredging in WA.  
149

## 150 *2.2. Predicting the impacts of dredging on life history characteristics*

151 When determining EWs for dredging for a particular taxon or sub-taxon, a model that  
152 accounts for external pressure (e.g. dredging pressures such as light reductions and suspended  
153 sediments, along with non-dredging pressures such as fishing, climate change) and  
154 vulnerability (LHC's, biological responses of plants) is required. The accuracy of the model  
155 is dependent both on how generalised the life history is for each taxon/sub-taxon, and on the  
156 accuracy of the prediction of spatial and temporal scales of dredging-related perturbations to  
157 the benthic environment. The model also depends on identifying feedback mechanisms  
158 between the dredging pressure and organism response. This becomes more complex when the  
159 timing of life history stages are considered, as these sensitive ecological processes differ  
160 between taxa such that the impact of dredging may vary across species within the same group  
161 in a particular area. However, this is not generally the case with seagrasses and macroalgae,  
162 and generalisations for these groups can be made based on season, sea temperature and light  
163 reaching the benthos. In the current review, we assess general vulnerability to dredging for  
164 marine invertebrates, seagrasses and macroalgae by assigning vulnerability scores based on  
165 their LHCs based on a literature review and expert elicitation drawn from workshop  
166 attendees. For the seagrasses, response to particular dredging pressures such as burial and  
167 decreases in light were also reviewed. With the exception of hard corals (see Erftemeijer et  
168 al., 2012 for review) data was limited for many of the taxa and groups examined in this  
169 review, often to studies conducted under extreme conditions that would be expected in areas  
170 of high impact, where effects are expected to be irreversible (EPA 2011). Our approach is  
171 nevertheless relevant and necessary for predicting vulnerable LHCs for these groups, and  
172 emphasizes the need for knowledge gaps to be addressed prior to EW classification.

### 173 2.3. Case study: Western Australia

174 Using WA as a case study, we demonstrate a general procedure for the consideration of EWs  
175 based on local knowledge, as well as the vulnerability scores established in this review.  
176 Representative species of invertebrates, seagrasses and macroalgae and, when possible, the  
177 timing of sensitive periods in their life histories in WA, were identified. Together with the  
178 previously established vulnerability scores, this information was used to assess when  
179 appropriate EWs may exist in this region.

180

## 181 3. Assessing the vulnerability of marine invertebrates to dredging

### 182 3.1. Potential impacts of dredging on invertebrates

183 Dredging and disposal can trigger ecological succession such that more opportunistic  
184 invertebrate species are likely to dominate shortly following a dredging event (Newell et al.,  
185 1998). Traits such as mobility, feeding mode, morphology and reproductive strategy  
186 contribute to the net vulnerability of a particular species to a dredging event (Essink, 1999).

187

#### 188 3.1.1. Mobile invertebrate species

189 Mobile invertebrates are generally less vulnerable than sessile taxa to sedimentation, as they  
190 are able to move to areas with less sediment accumulation or by more efficiently physically  
191 removing particles. Powilleit et al. (2009) measured mixed responses to heavy sedimentation  
192 (representative of conditions at a dredge disposal site) in the laboratory for Baltic Sea  
193 invertebrates, with survival rates of 0-33% depending on species and burial depth. Adult  
194 bivalves *Arctica islandica*, *Macoma balthica* and *Mya arenaria* and the polychaete *Nephtys*  
195 *hombergii* demonstrated a relatively high percentage of escape (restored contact with surface  
196 water) after burial in 32-41 cm of sediment. Some polychaetes (*Bylgides sarsi*) managed  
197 escape from 16 cm of sediment, while others (*Lagis koreni*) did not migrate (Powilleit et al.,  
198 2009). Mobility alone does not indicate that these groups are resistant to dredging as certain  
199 critical life stages are still susceptible to several indirect effects of sedimentation. For  
200 example, juveniles of the blackfoot abalone (*Haliotis iris*) in New Zealand are not directly  
201 impacted by sedimentation, but have been observed to reorientate themselves during  
202 sedimentation events from a horizontal position underneath the cobbles (a predation refuge)  
203 to an upright position on the sides of the cobbles, increasing their vulnerability to predation

204 (Chew et al., 2013). These examples highlight the importance of understanding the response  
205 of certain life stages of individual species to sedimentation on a location-specific basis as  
206 well as how ecological interactions may be modified under such conditions.

207

### 208 3.1.2. Sessile invertebrate species

209 Sessile invertebrates are particularly vulnerable to sedimentation because they are generally  
210 unable to reorientate themselves to mitigate a build-up of particulates. Some sessile taxa,  
211 including species of sponges and bivalves, have the capacity to filter out or to physically  
212 remove particulates, however this can be metabolically costly and unsustainable (Gerrodette  
213 and Flechsig, 1979; Cortés and Risk, 1985; Aldridge et al., 1987; Roberts et al., 2006, Pineda  
214 et al. 2016). The impact of sedimentation on sessile invertebrates depends on a range of  
215 additional factors, including the duration of exposure (Suedel et al., 2014) and proximity to  
216 dredging activities (EPA 2011). Morphology plays a critical role since upright morphologies  
217 are generally more resistant to burial than encrusting forms. Indeed, studies on the  
218 sedimentation and burial of rocky sublittoral sponge communities have measured a decrease  
219 in morphological diversity with increased sedimentation (Carballo, 2006).

220

221 A recent review on sponge-sediment relationships in Western Australia revealed a series of  
222 beneficial interactions of sediment-adapted marine sponges with sediments through  
223 morphological adaptations, including sediment skeletons and surface crusts (reinforcement),  
224 stalks and fistules (elevation above sediments), and spicule tufts and root-systems  
225 (anchoring) (Schönberg, 2016). Similarly, many sea whips and other gorgonian species along  
226 the Florida Gulf Coast are relatively resistant to dredge-related sedimentation due to their  
227 morphology, which resists the build-up of sediment (Marszalek, 1981).

228

229 Diet and feeding mode are also important in driving species vulnerability to sedimentation  
230 and light attenuation, especially in sessile species. Sedimentation can be particularly  
231 detrimental for suspension feeding organisms since suspended particles can be mistaken for  
232 food (Bell et al. 2015). In addition, the mechanical or abrasive action of suspended sediments  
233 may be harmful to suspension feeders, clogging their feeding apparatus and impairing  
234 respiratory and excretory function (Sherk, 1972). Several sessile invertebrate taxa such as  
235 sponges possess photosynthetic symbionts (Lemloh et al. 2009; Keesing et al. 2012) and light  
236 attenuation has the potential to disrupt these relationships (Roberts et al., 2006).



237

## 238 3.1.3. Reproductive and developmental strategy

239 A species' reproductive strategy, reproductive season and developmental strategy are also  
240 major factors contributing to their vulnerability. For example, semelparous organisms, which  
241 have a single reproductive episode in a life-cycle, would be expected to be more vulnerable  
242 to disturbances than iteroparous organisms, which may reproduce multiple times in a  
243 lifecycle (Roberts and Hawkins, 1999). Similarly, the risk for adverse effects of dredging  
244 during reproductive periods is greater for invertebrates with a discrete annual spawning  
245 period compared to those with multiple protracted spawning events occurring throughout the  
246 year and outside the period of dredging. Some species have well studied and predictable  
247 spawning periods which enable discrete environmental windows in some areas. Most notably,  
248 most species of scleractinian corals on the Great Barrier Reef have a very discrete spawning  
249 period associated with the lunar cycle in spring (Babcock et al., 1986) Species from other  
250 phyla have also been observed spawning in concert with the corals during these annual  
251 autumn spawning events (Babcock et al., 1992). In addition the reproduction biology for  
252 some commercially fished crustaceans and molluscs is well known including in Western  
253 Australia (e.g. Caputi et al. 1998), but for many taxa and in many locations where dredging  
254 may occur, reproductive periodicity of invertebrates has been studied very little.

255 Developmental strategy is also important. Brooding invertebrate species, with a limited  
256 capacity for dispersal, are generally more vulnerable than those with planktonic larval stages  
257 that may facilitate the colonisation of new, undisturbed habitats (Roberts and Hawkins,  
258 1999). However, meroplanktonic species entering or remaining inside an area being dredged  
259 may be highly vulnerable to the mechanisms of dredging since high concentrations of  
260 suspended sediments and elevated sedimentation rates can impair larval swimming and  
261 inhibit larval settlement and recruitment (Wilber and Clarke, 2001; Wilber et al. 2005). In the  
262 water column, oyster larvae can tolerate suspended particulate matter concentrations of up to  
263 400-800 and 2200 mg L<sup>-1</sup>, respectively (Wilber and Clarke, 2001). However, once ready to  
264 settle, larvae may have difficulty attaching to substrata that are covered in a layer of fine  
265 sediment (Wilber et al. 2005). Indeed, the deposition of sediment on mussel beds hinders  
266 settlement, attachment and survival of mussel larvae (Bender and Jensen, 1992, Wilber and  
267 Clarke 2010). Similarly, in the Florida Keys, the spiny lobster, *Panulirus argus*, has reduced  
268 rates of settlement in heavily silted areas (Herrnkind et al., 1988). In contrast, oyster larvae  
269 can tolerate thin layers of sediment (up to 1 mm). In the early stages of attachment, the

270 deposition of fine sediments is likely to have a negative effect on recruit survival, whereas  
271 following this period juvenile oysters can tolerate sediment deposition of 2-3 mm, but >5 mm  
272 is likely to have negative effects (Wilber and Clarke, 2001). Fine sediments may also create a  
273 boundary layer for gas transfer, facilitating the formation of sulphides and creating anoxic  
274 conditions (Salomons, 1985), which may inhibit the growth of attached organisms or cause  
275 mortality (Essink, 1999).

### 276 *3.2. Identifying key life history characteristics of invertebrates and assigning vulnerability* 277 *scores*

278 There is potential for significant negative effects from dredging operations if conducted  
279 during the key periods of larval release, settlement and recruitment. Identifying the timing of  
280 these ecologically sensitive periods on a species-specific basis is therefore important for  
281 environmental window modelling for a particular location. A generalised summary of LHCs  
282 that may be vulnerable to dredging for various life history stages is shown in Table 1.  
283 Detailed information on the LHCs of major invertebrate taxa is shown in Appendix A. We  
284 created sub-categories for each phylum, based on morphology and reproductive strategy, as  
285 these are major factors contributing to a species' vulnerability to sedimentation. This  
286 information was then used to assign a general vulnerability score to each taxon or taxon  
287 subset listed in Appendix A.

288

## 289 **4. Assessing the vulnerability of seagrasses to dredging**

### 290 *4.1. Potential impacts of dredging on seagrasses*

291 Seagrasses are highly sensitive to changes in water quality, sediment loading, and other  
292 inputs that accumulate as a result of the modification of watersheds and coastal water bodies  
293 (Dennison et al., 1993). Therefore, seagrasses are useful for identifying critical environmental  
294 thresholds that may be triggered by dredging operations for other organisms. Given the  
295 widespread distribution and significant environmental and economic value of seagrass  
296 ecosystems (Orth et al., 2006), these organisms take priority for protection within dredging  
297 management practice (Waycott et al., 2009).

298

299 Seagrasses can be affected by dredging in several ways. They can be directly affected at the  
300 dredge and disposal sites, when they are often physically removed or buried, or indirectly by

301 changes in water quality or bathymetric changes which may sometimes occur as a result of  
302 dredging activities (Erftemeijer et al., 2006). Seagrasses are also affected by the increased  
303 turbidity, resulting in reductions in light available for photosynthesis, and increased levels of  
304 sedimentation, which can result in significant negative effects on seagrass shoot density, leaf  
305 biomass, physiology and productivity (Erftemeijer et al., 2006).

306

307 The ability of seagrasses to resist and recover from disturbances caused by dredging is  
308 species-specific and related to a number of LHCs (Table 1(B)). Recently, Kilminster et al.  
309 (2015) summarised seagrass vulnerability to disturbance by grouping species into three  
310 categories based on their LHCs: 1) *Persistent* species are defined as those with long turnover  
311 times, that are slow to reach sexual maturity and with less investment in sexual reproduction  
312 such that the presence of a seed bank is rare. Persistent species are more resistant to  
313 disturbance but take longer to recover than colonising species; 2) *Opportunistic* species share  
314 traits with the previous and next classifications, with the ability to colonise quickly, produce  
315 seeds and to recover from seed when necessary (Kilminster et al., 2015); and 3) *Colonising*  
316 species are seagrasses with short ramet turnover times, that are quick to reach sexual maturity  
317 and display a high investment in sexual reproduction to produce seeds, usually resulting in  
318 the presence of a seed bank. Species within this group generally have a limited resistance to  
319 disturbance but have the ability to recover quickly. We use these classifications for assessing  
320 seagrass vulnerability as LHCs play a large role in determining the vulnerability (ability to  
321 resist and recover from disturbance; with lower resistance and recovery rates leading to high  
322 vulnerability) of a particular species to dredging (Kilminster et al., 2015).

#### 323 4.2. Identifying key life history characteristics of seagrasses and assigning vulnerability 324 scores

##### 325 4.2.1. Persistent seagrass species

326 Persistent species generally have high resistance to dredging-disturbance but slow rates of  
327 recovery once they have suffered losses. Within the case study area, there are three genera of  
328 persistent seagrasses, one temperate (*Posidonia*) and two tropical (*Thalassia* and *Enhalus*).  
329 *Posidonia* species are grouped within the *persistent* classification by Kilminster et al. (2015)  
330 based on their LHCs. These species are found in temperate and sub-tropical regions of  
331 Australia. Water quality and other environmental conditions in these habitats follow a strong  
332 summer-winter seasonal cycle. Light levels and temperature in these regions are higher

333 during summer and levels of suspended sediment are greater in winter due to increases in  
334 riverine input and storm-driven resuspension. The capacity of large-bodied, foundation  
335 seagrasses like *Posidonia* spp. to survive short-term reductions in light after a dredging event  
336 is high, but after extensive periods of shading these species tend to experience significant loss  
337 of biomass and shoot density, with minimal recovery. *Posidonia* spp. response and recovery  
338 following periods of reduced light, burial and sedimentation is species-specific and  
339 dependent on many additional factors such as the extent and duration of light reduction, as  
340 well as the depth of burial (Table 2). Indeed, *Posidonia oceanica* and *Posidonia coriacea*  
341 have the lowest light requirements 7-8 % surface irradiance (SI) (Duarte, 1991) and 8 %SI  
342 (Westphalen et al., 2004), respectively, while *Posidonia sinuosa*, *Posidonia australis* and  
343 *Posidonia angustifolia* have minimum light requirements of 7-24 %SI (Duarte, 1991;  
344 Westphalen et al., 2004), 10 %SI (Fitzpatrick and Kirkman, 1995), and 7-24 %SI (Duarte,  
345 1991), respectively. Generally, *Posidonia* species shows very slow or no recovery following  
346 impacts from prolonged reductions (198 days) in light availability (Collier et al. 2009).

347

348 Species within the tropical genera *Thalassia* and *Enhalus* are relatively large-bodied and  
349 slow-growing and thus also classified as *persistent* (Kilminster et al., 2015). These seagrasses  
350 display some tolerance to sedimentation (Waycott et al., 2007; Cabaço et al., 2008). For  
351 example, *Thalassia testudinum* and *Thalassia hemprichii* displayed 50% mortality under 5cm  
352 of sediment (Suchanek, 1983), while only 20% mortality of *Enhalus acoroides* was observed  
353 after 10 months burial under 16 cm of sediment (Cabaço et al., 2008). *Thalassia* spp. are  
354 negatively affected by dredged sediment plumes due to the reduced light availability  
355 associated with turbidity. *T. testudinum* in Corpus Christi Bay, Texas, experienced 99%  
356 mortality after 490 days under 14 %SI, and 100% mortality after 200 days under 5 %SI (Lee  
357 and Dunton, 1997). This mortality was preceded by reductions in leaf productivity, and  
358 indices such as shoot density, blade width, leaf growth, chlorophyll a:b and blade chlorophyll  
359 content. These physiological and morphological changes may be important early indicators of  
360 chronic stress from light reductions associated with dredging (Lee and Dunton, 1997).  
361 *Enhalus* spp. naturally occurs in highly turbid environments (Kiswara et al., 2005; Unsworth  
362 et al., 2012) as is therefore likely to be more resistant to *Thalassia* spp. to reduced light  
363 availability. *Thalassia* spp. recover relatively slowly following disturbance, taking several  
364 years to recover from vessel-related injury (3.5 - 4.1 years for propeller scars and up to 7.6  
365 years for artificial cuts) (Dawes et al., 1997). It is therefore critical that *Thalassia* spp.

366 meadows are not damaged beyond their threshold of recovery, as complete recovery and  
367 regrowth of a damaged meadow may take many years.

368

#### 369 4.2.2. Opportunistic seagrass species

370 Opportunistic seagrass genera (e.g. *Amphibolis*, *Zostera*, *Cymodocea*, *Syringodium*) have  
371 variable resistance to dredging. For example, the moderately fast growing genus *Amphibolis*  
372 is generally more resilient to sedimentation and burial than smaller genera with shorter life  
373 spans. Indeed, *Amphibolis* growth rates were unaffected following burial in 10 cm of aerobic  
374 sediment along the Adelaide coast (Clarke, 1987). However, other opportunistic genera have  
375 limited resilience to burial. *Zostera* spp. have shown limited resilience to burial (70 - 90%  
376 mortality under 2-4 cm sediment) (Mills and Fonseca, 2003; Cabaço and Santos, 2007 ), and  
377 large losses of *Zostera tasmanica* and *Zostera muelleri* were attributed to dredging and  
378 sediment build up on leaves (Kirkman, 1978, Clarke and Kirkman, 1989). Similarly, sudden  
379 burial under 5 cm of sediment resulted in 90% mortality in *Cymodocea nodosa* after 35 days  
380 (Marba and Duarte, 1994). However, *Cymodocea serrulata* and *Syringodium isoetifolium*  
381 were able to withstand burial under 4 cm for 27 days, but responded adversely to burial  
382 depths greater than 8 cm with large reductions in above and below ground biomass and shoot  
383 density (Ooi et al., 2011). Both species also benefitted from the presence of an intact  
384 rhizome, such that clonal integration is important for the persistence of these species  
385 following a burial event (Ooi et al., 2011), likely due to the sharing of resources between  
386 neighbouring individuals (ramets) (Marba et al., 2006).

387

388 Most opportunistic seagrass genera have high rates of recovery following disturbance.  
389 *Amphibolis griffithii* meadows have been shown to recover within 10 months from shading  
390 experiments mimicking 3-month long dredging scenarios despite above-ground biomass  
391 losses of up to 72% (McMahon et al., 2011). However, recovery was not observed following  
392 longer periods of shading (6-9 months, McMahon et al., 2011). *Cymodocea* species have the  
393 potential to recover from periods of eutrophication and/or light deprivation once conditions  
394 improve. For example, a *Cymodocea nodosa* meadow in a Mediterranean lagoon decreased  
395 by 49% in cover from the early 1970s to the early 1990s due to heavy rainfall, dredging and  
396 eutrophication, but subsequently increased by 42% from the early 1990s to 2013, initiated by  
397 improved catchment management and termination of dredging, showing capacity for  
398 recovery (Garrido et al., 2013). Seagrass species within the *Zostera* genus also show a

399 relatively high capacity for recovery, both from seed reserves and clonal growth. In the  
400 tropics, *Z. muelleri* recovered completely within two years following a flood-related loss of  
401 95% of intertidal seagrasses in the Great Sandy Strait, Queensland, Australia, with recovery  
402 facilitated by seed banks in sediments (Campbell and McKenzie, 2004). Thus, maintenance  
403 of seed banks may be critical to the recovery of damaged *Zostera* spp. beds and dredging  
404 operations timed after seed release are more likely to facilitate natural re-growth from seed  
405 reserves. However, in other meadows of the same species recovery from loss may be nearly  
406 exclusively from clonal growth (Rasheed 1999) with prognosis for recovery poor if the entire  
407 standing crop is lost. In these circumstances maintenance of the adult population may be  
408 more critical than protecting flowering and seed production (Rasheed 1999) and emphasise  
409 the requirement for local knowledge of meadow life history and differences that can occur  
410 even within the same species before application of EW's.

#### 411 4.2.3. Colonising seagrass species

412 Colonising seagrass genera (e.g. *Halodule*, *Halophila*) have low resistance to short term  
413 pulses of increased turbidity and sedimentation in comparison to larger-bodied persistent or  
414 opportunistic species. Sedimentation and burial (4-8 cm depth) of a mixed seagrass meadow  
415 that included *Halodule uninervis* resulted in reductions in shoot density and limited recovery  
416 after 10 months (Duarte et al., 1997). However, *Halodule wrightii* can survive in light  
417 conditions between 5-30 %SI depending on the depth, water colour and natural turbidity  
418 fluctuation (Erftemeijer et al., 2006). *Halophila ovalis* has been reported to have a relatively  
419 low tolerance to burial (Vermaat et al., 1997). However, *Halophila ovalis* and *Halodule*  
420 *uninervis* were able to withstand burial under 4 cm for 27 days, though burial depths greater  
421 than 8 cm resulted in large reductions in biomass (Ooi et al., 2011). Furthermore, clonal  
422 integration is less important for the recovery of these genera following burial than for some  
423 of the other tropical seagrasses (Ooi et al., 2011), possibly due to their smaller size and  
424 limited communication and resource sharing between ramets (Marba et al., 2006).

425

426 The relatively fast growth rates and high rates of reproduction characteristic of *Halophila*  
427 spp. and other colonising species can decrease their vulnerability to disturbance (Demers et  
428 al., 2013; Kilminster et al., 2015). Seagrasses within the genus *Halophila* and other  
429 *colonising* species grow quickly from a stored seed bank and may therefore re-colonise  
430 dredged areas through seed dispersal (Kilminster et al., 2015). As such, *Halophila* spp. can  
431 generally recover following sedimentation and burial if seed banks are present (Hovey et al.,

2015), unless the seeds are buried under too much sediment, preventing the hypocotyl from penetrating the sediment surface (Birch, 1981). In addition, 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 size. *Halophila ovalis* shows acclimation potential to light levels below their minimum light requirements, but only for 3-5 days, after which growth rates are reduced (Longstaff et al., 1999a,b). Recovery was possible for this species if light levels were restored within 9 days, but periods of low light exceeding 15 days were associated with an exponentially greater risk of mortality, with 100% mortality occurring after 30 days of shading (Longstaff et al., 1999a). Acclimation to chronic low light conditions in *Halophila ovalis* in Singapore waters was found to reduce its resilience to further (short-term) sediment disturbances (Yaakub et al., 2014). Similarly, the capacity for recovery after a loss is high in *Halodule wrightii*, with documented recovery of 2000 ha of seagrass in Tampa Bay, Florida, USA, due to improved water conditions (Johansson, 2002).

447

#### 4.2.4. Implications for selecting environmental windows for seagrasses

The use of EWs prior to dredging can be important for ensuring seagrass recovery following a dredging event. EWs must take seagrass biology, phenology and environmental seasonality for each species at each site into account. Life span, growth rate and reproductive strategy are critical aspects of plant biology that contribute to the overall vulnerability of seagrass species (Table 1(B)), while important phenological considerations include flowering, fruiting and sensitive life-history stages (e.g. seedling vs. mature plant stages). Furthermore, the most sensitive period in the life cycle may depend on the life-history strategy of a particular species. For example, for persistent species the adult plant would take priority for protection, whereas for colonising species, periods of reproduction and production of seedbanks would be more vulnerable and take priority. A summary of characteristics that contribute to seagrass vulnerability to dredging is given in Table 1(B). In general, slow-growing seagrasses that take longer to reach sexual maturity and do not form sediment seed banks will have a higher vulnerability than fast-growing seagrasses with short turnover times, shorter periods to reach sexual maturity, and sedimentary seed banks. In addition, seasonal environmental cycles such as light, temperature, tidal cycles (i.e. periods of exposure), storms and other periods of high swell/wave energy must be considered. Assessing the temporal cycle of plant phenology

464

465 (reproductive or vegetative) together with temporal cycles in potential environmental  
466 stressors can reveal periods when the environmental impacts of dredging will be increased.

467

## 468 **5. Assessing the vulnerability of macroalgae to dredging**

### 469 *5.1. Potential impacts of dredging on macroalgae*

470 Biological traits such as growth rate, life span and reproductive strategy contribute to the  
471 vulnerability of macroalgal species to dredging (Table 1(C)). In terms of phenology, sensitive  
472 periods in the life history cycle should be considered (e.g. gametophyte vs. sporophyte stages  
473 for macroalgae genera with heteromorphic life histories). We split our discussion on  
474 macroalgae into classifications based on widely accepted functional groups from Steneck and  
475 Dethier (1994). These functional groups split macroalgae up based on organismal features  
476 such as anatomy, morphology and productivity. We restrict our discussion to groups that  
477 contain taxa where more information regarding responses to dredging are available.

#### 478 5.1.1. Leathery Macrophytes

479 The 'leathery macrophyte' group includes genera such as *Sargassum* and *Ecklonia* that are  
480 major habitat formers in temperate and tropical reefs, and have ecologically important roles  
481 such as habitat and food provision (Steneck et al., 2002). Brown algae within the genus  
482 *Sargassum* are common in nearshore ecosystems, and are thought to have an advantage in  
483 higher sediment environments due to their abundance in turbid, inshore reef habitats (e.g. on  
484 the Great Barrier Reef). Schaffelke (1999) observed an increase in *Sargassum* spp. growth  
485 rates of up to 180% when particulate matter was present on the thallus surface, potentially  
486 due to the creation of a nutrient-rich boundary layer. *Sargassum* spp. appear to be resistant to  
487 the negative effects of sedimentation if it is already established in a system, but observed  
488 increases in *Sargassum* spp. abundance may not be directly related to the sedimentation  
489 event, and instead to a release from competition or predation. In contrast, increased  
490 sedimentation levels in a fringing reef environment led to significantly decreased rates of  
491 recruitment, growth, survival and vegetative regeneration in *Sargassum microphyllum* (Umar  
492 et al., 1998). Successful settlement of brown algae such as kelps on hard bottom substrata is  
493 inhibited by sediment, with a direct relationship between settlement success and the thickness  
494 of the sediment for some algal species (Chapman and Fletcher, 2002). Thus, the effects of  
495 sedimentation on *Sargassum* spp. are variable. Due to the increased sensitivity of leathery



496 macrophytes to sedimentation during reproductive and recruitment phases, it would be  
497 beneficial to avoid these periods for dredging. Algae in the temperate genus *Ecklonia*,  
498 another habitat-forming leathery macrophyte, have a heteromorphic life history wherein the  
499 large, conspicuous plant (the sporophyte) alternates with a small, filamentous gametophyte,  
500 which is the site of sexual reproduction. Erosion, or tissue sloughing, of *Ecklonia* spp.  
501 sporophytes occurs in autumn (de Bettignies et al. 2013), and survival and growth of the  
502 gametophyte is tolerant of high and variable summer temperatures (Mohring et al. 2014),  
503 such that an autumn-winter EW is clearly preferred for these large macrophytes.

504 The detailed phenology of most brown algae is poorly known but appears to be temperature  
505 dependent (Kendrick and Walker, 1994). An annual cycle of vegetative growth, reproduction  
506 and senescence in *Sargassum* spp. is often reported, but its timing varies between temperate  
507 and tropical regions. In general, *Sargassum* spp. can be most abundant during the warmest  
508 part of the year in temperate regions in Australia (Kendrick and Walker, 1994); or most  
509 abundant when temperatures are lowest in the tropics (De Wreede, 1976; Ang 2007). Yet in  
510 tropical environments like Ningaloo Reef in Western Australia, growth and reproduction of  
511 *Sargassum* spp. are greatest in the warmer months (Fulton et al. 2014). However, tides also  
512 play a role as noted in the Philippines, where phenological patterns of two *Sargassum* spp.  
513 populations experienced die-back during a period of prolonged exposure related to the lowest  
514 tide of the year (Ang 1985).

515

### 516 5.1.2. Siphonous algae

517 The functional group 'siphonous algae' consist entirely of green algae from the order  
518 Bryopsidales. The effects of dredging and sedimentation on siphonous algae are similar to the  
519 leathery macrophytes. Low levels of sedimentation are unlikely to inhibit algal growth but  
520 may affect recruitment, survival and vegetative regeneration. Furthermore, invasive  
521 Chlorophytes in the Mediterranean Sea, such as *Caulerpa racemosa* (now *C. cylindracea*),  
522 appear to be more resistant than native species to sedimentation events, thus benefitting from  
523 such disturbances (Piazzi et al., 2005). Calcareous green algae within the genus *Halimeda*  
524 have a noted tolerance to lower light levels (Hillis-Colinvaux, 1986) and may thus be more  
525 resistant to the increases in turbidity associated with dredging than other genera with higher  
526 light requirements.

527 As with the brown algae, the phenology of most green algae is poorly known. In the  
528 Caribbean, Clifton and Clifton (1999) noted a broadly seasonal peak of reproductive activity

529 in green algae that coincided with the annual shift from the dry to the wet season in Panama  
530 (March – June; a period of increased solar radiation). In Australia, Price (1989) recorded  
531 active growth of most species during autumn, winter and spring, whereas smaller groups  
532 were restricted to winter and spring, and others to summer. As such, generalities with respect  
533 to siphonous algae phenology cannot be made without considerable further study.

### 534 5.1.3. Crustose Coralline Algae

535 Crustose coralline algae (CCA) are ecologically important in the habitats in which they  
536 occur, contributing to carbonate accretion, structural complexity and facilitating the  
537 settlement and recruitment of many other taxa (Nelson, 2009). As such, their response to  
538 sedimentation and burial will have major ecological ramifications on a community-wide  
539 scale. The distribution of CCA on the Great Barrier Reef has shown strong links to the  
540 sedimentation environment. Near-shore reefs exposed to higher sedimentation had a much  
541 lower abundance of CCA, and abundance increased from the middle to the outer shelf with  
542 increases in water clarity, reef slope and a decrease in sedimentation (Fabricius and De'Ath,  
543 2001). Despite distribution patterns suggesting that CCA are sensitive to sedimentation, CCA  
544 can survive long periods of burial by sloughing off epithelial cells such that underlying tissue  
545 can survive after the sediment is removed (Keats et al., 1997). Despite their resistance to the  
546 negative effects of burial, CCA are sensitive to the reductions in light associated with  
547 sedimentation (Riul et al., 2008). In contrast, foliose species of red algae are relatively  
548 tolerant to reductions in light. For example, the shade-adapted red alga *Anotrichium crinitum*  
549 has minimum light requirements of 1.49–2.25  $\mu\text{mol photons m}^{-2}\text{s}^{-1}$  and 0.12–0.19  $\mu\text{mol}$   
550  $\text{photons m}^{-2}\text{d}^{-1}$  for the initiation of photosynthesis and growth, respectively (Pritchard et al.,  
551 2013). This group can also tolerate sub-optimal light conditions for up to five days without  
552 losing biomass (Pritchard et al., 2013).

553 Rhodophyta species show considerable variation in periodicity of growth and reproduction  
554 (Price, 1989), such that no general pattern is evident. Maggs and Guiry (1987) suggested that  
555 temperature, photoperiod, light quality and irradiance are the most important environmental  
556 factors regulating macroalgal phenology, although temporal variation in nutrient levels,  
557 grazing pressure, wave action and sand scour may also be important in some communities.  
558 Rhodophyte species with heteromorphic life histories including crustose or filamentous  
559 phases are often prevalent on mobile substrata and are able to withstand severe conditions,  
560 although existing studies are limited.

561 5.2. *Identifying key life history characteristics of macroalgae and assigning vulnerability*  
562 *scores*

563 As with seagrasses, environmental windows for macroalgae should account for plant  
564 phenology, sensitive periods in the life history cycle (e.g. gametophyte vs. sporophyte stages  
565 for some macroalgae) as well as annual cycles in environmental conditions. However,  
566 phenology for many taxonomic and functional groups of macroalgae remains poorly known,  
567 limiting our capacity to determine vulnerability and identify potential EWs. In general, slow-  
568 growing macroalgae that are longer-lived will be more vulnerable than faster-growing  
569 macroalgae that have shorter lifespans.

570 **6. Case study: the potential for environmental windows in Western Australia**

571 The selection of effective EWs is highly dependent on the particular habitat and species  
572 present. These may be highly diverse, with correspondingly diverse LHCs and variable  
573 vulnerabilities to disturbance. Thus, the first step in the selection of EWs for dredging is to  
574 assess the ecological, social and economic ‘value’ of the species present in order to prioritise  
575 protection. This assessment should be made in tandem with robust estimates of the probable  
576 spatial and temporal scales of dredging-induced sediment release and sedimentation  
577 superimposed on exposures due to other sources (e.g. seasonal frequencies of freshets and  
578 storms). Finally, the vulnerability of these species must be assessed based on their LHCs and  
579 sensitivity to environmental change.

580 6.1. *Environmental windows for marine invertebrates in Western Australia*

581 A vulnerability index based on LHCs and timing of reproduction for known species of  
582 marine invertebrates in Western Australia is shown in Appendix B. This information is  
583 incomplete, as the timing of reproductive events for many of these species has not been  
584 investigated.

585 6.1.1. Temperate invertebrates

586 In temperate Western Australian waters information on the reproductive periodicity of a large  
587 number of species exists on which to base a view about appropriate EWs. Many species of  
588 cnidarians, molluscs, crustaceans and echinoderms undergo gametogenesis in the spring and  
589 spawn (or planulate) in summer or early autumn (e.g. the corals *Pocillopora damicornis*  
590 Stoddart and Black 1985 and *Goniastrea australensis*, Crane 1999; the molluscs *Turbo*  
591 *torquata*, Joll 1980; the rocklobster *Panulirus Cygnus*, Chittleborough and Thomas 1969; the

592 prawn *Penaeus latisulcatus*, Penn 1980; the sea star *Archaster angulatus*, Keesing et al.  
593 2011; and the sand dollar *Peronella lesueuri*, Yeo et al. 2015). Different coral species do not  
594 spawn synchronously in south-western Australia as they do in some places but the species  
595 that have been studied have their spawning limited to between February and April (Crane  
596 1999). Thus, based on the available information, a potentially effective EW for dredging on  
597 invertebrate dominated reefs in temperate Western Australia appears to be during the winter  
598 months (June – Sept., Figure 1), when the least number of known species are undergoing  
599 periods of spawning and recruitment. However, there are exceptions to this pattern of  
600 summer spawning among temperate Western Australian molluscs, for example the  
601 commercially important abalone *Haliotis roei* spawns in July and August in south-western  
602 Australia (Wells and Keesing 1989 and the chiton *Acanthopleura hirtosa* spawns in April and  
603 June (Wells and Sellers 1987) and the trochid *Cantharidus pulcherrimus* spawns from March  
604 to April (Wells and Keesing 1987). Some species have more than one spawning period  
605 during the year (e.g. the limpet *Patelloida nigrosulcata* spawns in winter and spring (Wells  
606 and Keesing 1988) which is helpful to identifying EWs.

#### 607 6.1.2. Tropical invertebrates

608 In north-western Australia, most species of scleractinian corals are known to spawn  
609 synchronously after sunset on an ebbing neap tide during a discrete and predictable annual  
610 window in late March (autumn) (Simpson 1985; 1991; Simpson et al. 1993; Babcock et al.  
611 1994) although spawning of some species also occurs in spring or early summer in parts of  
612 the Pilbara and Kimberley, consistent with that on the Great Barrier Reef (Rosser and Baird  
613 2009; Baird et al. 2011; Stoddart et al. 2013) leading to a suggestion that activities to which  
614 coral gametogenesis, spawning and settlement also be avoided during that period (Baird et al.  
615 2011). Species from other phyla have also been observed spawning in concert with the corals  
616 during annual autumn spawning events on the Great Barrier Reef (Babcock et al., 1992) and  
617 as such this might also be expected to occur in Western Australia. Indeed Simpson et al.  
618 (1993) recorded unidentified polychaetes spawning at the same time as *Acropora* corals at  
619 Ningaloo Reef in Western Australia. Those same authors have also observed a polychaete  
620 (*Eunice* spp.) releasing a bright red epitoke as well as sea stars and sea urchins spawning  
621 coincident with corals at Ningaloo (R. Masini, pers. obs; C. Simpson, pers. comm.). With the  
622 exception of commercially important invertebrate species (e.g. *Penaeus latisulcatus* spawns  
623 year round in north-western Australia, Penn, 1980), the reproductive periodicity of species  
624 that are not synchronised with scleractinians are not as well known. However, the heart

625 urchin *Breyntia desorii* spawns in June in the Kimberley (Keesing and Irvine 2013) and  
626 occasional observations that indicate spawning times of other tropical Western Australian  
627 marine invertebrate species outside of the coral spawning season are made but are rarely  
628 reported in the formal literature (e.g. the sea stars *Protoreaster lincki* and *Protoreaster*  
629 *nodulosus* in the Pilbara region in November, Keesing pers. obs). Thus not all tropical marine  
630 invertebrates in Western Australia spawn within the same EW as corals and there is an  
631 immediate need to establish a more synoptic picture of EWs in Western Australia based on  
632 new biological and ecological studies. In the absence of this information, EWs during the  
633 neap tide in autumn and winter, established to reduce dredging related turbidity generation,  
634 would protect sensitive life stages of some important taxa from turbidity-related stress in  
635 northwest Western Australia.

636

## 637 6.2. Environmental windows for seagrasses in Western Australia

638 The overall vulnerability of seagrasses to dredging is primarily determined by LHCs and  
639 physiological responses to light reduction (Appendix C). LHCs that may influence  
640 vulnerability to dredging are shown in Appendix D, and detailed phenological information  
641 for major Western Australian seagrass genera is shown below. Given that the timing of  
642 reproduction and recruitment, as well as sensitivity to light reductions are highly species-  
643 specific, and in some cases location specific (Table 3, Appendix C), EWs selected for the  
644 protection of seagrass beds in Western Australia will depend on the species present in a  
645 particular location.

### 646 6.2.1. Temperate seagrass meadows

647 In the persistent seagrass, *Posidonia* species flower from autumn through to spring and fruit  
648 over early summer. Fruit, containing a single viable seed, are released continuously during 2-  
649 3 weeks in early summer and may float for up to a week before the fruit dehisces the seed and  
650 the seed is deposited (Cambridge, 1975). *Posidonia* spp. seeds have significant reserves and  
651 seedlings establish and grow on those reserves for 6 months - 1 year (Statton et al., 2013). In  
652 adult plants, carbohydrate stores are typically much larger in summer than winter, indicating  
653 less light limitation (Collier et al., 2009). Furthermore, leaf extension rates (Collier et al.,  
654 2007) and root length (Hovey et al., 2012) are greater in summer. As such, avoidance of  
655 intensive dredging activities during the summer months (Oct. – April) could be considered as

656 EWs for *Posidonia* in order to protect periods of seed release and dispersal, as well as high  
657 productivity and growth.

658

659 The opportunistic, *Zostera polyclamys (tasmanica)* reproductive structures have been  
660 observed in September and mature seed bearing spathes have been observed during summer  
661 (Nov. – Jan.) (Kirkman, 1999, Campey et al. 2002). In Victoria, Australia, the greatest rates  
662 of *Zostera* spp. leaf and areal production have also been measured during summer and late  
663 spring (Bulthuis, 1983; Bulthuis and Woelkerling, 1983). Thus, EWs that avoid intensive  
664 dredging activities in close vicinity to seagrass areas during spring and early summer in  
665 temperate environments are likely to be beneficial for this genus.

666

667 The opportunistic, *Amphibolis* species flower during the Austral autumn, between May and  
668 October. The seeds germinate on the adult plant and are released as mature seedlings between  
669 November and June, and seedlings are present year round. *Amphibolis* spp. meadows are  
670 most productive during summer (Dec. – Feb.) and reduced but relatively constant for the  
671 remainder of the year (Walker and McComb, 1988), such that reserves are likely to be  
672 established during summer. Shading of *Amphibolis griffithii* meadows reduces leaf and root  
673 biomass more in late summer than late winter (Lavery et al., 2009). Therefore, it is possible  
674 that dredging in the months leading up to flowering (i.e. during autumn) could reduce  
675 carbohydrate reserves and flowering, while dredging in summer could lead to declines of  
676 existing *Amphibolis* spp. meadows. Overall, the optimal period for dredging in areas  
677 dominated by temperate seagrass meadows in Western Australia is during the winter months.

#### 678 6.2.2. Tropical seagrass meadows

679 For colonising *Halophila* species, such as *Halophila decipiens*, cycling between active  
680 growth and dormant seed bank stages in the life history is triggered by environmental cues.  
681 Therefore, periods exist when dredging activity may have little impact on *Halophila* species.  
682 For example, the natural light climate in the Kimberley region, Western Australia, involves  
683 fluctuations of low light (10-0 %SI) in the wet season (Nov. – April), and higher light levels  
684 (20-2 %SI) during the dry season (May – Oct.) (Hovey et al. 2015). The lifecycle of *H.*  
685 *decipiens* follows light availability, with dormant seed dispersal stages during the darker wet  
686 season, and seedling growth, meadow development and gamete production occurring during  
687 the lighter dry season (Hovey et al. 2015). As such, vulnerability to sedimentation and  
688 reduced light is low during the dormant seed dispersal stage during the wet season, and

689 presents an EW to reduce the impact of dredging. In contrast, intensive dredging activities  
690 could have major impacts on this species during the dry season in this region, as has been  
691 seen for the species on the east coast of Australia (York et al. 2015) when the plants rely on  
692 higher light levels to stimulate germination of the seed bank, meadow development flowering  
693 and seed production. While this window may be appropriate for colonising seagrass species  
694 the same may not hold true for opportunistic and persistent tropical species that have a less  
695 pronounced seasonality in life history and a higher reliance on the adult phase to confer their  
696 resilience to impacts.

697

### 698 6.3. Environmental windows for macroalgae in Western Australia

699 Vulnerability scores based on LHCs for major Western Australian macroalgal genera are  
700 shown in Appendix E. Based on these scores and the timing of reproduction and recruitment  
701 for these groups (Table 3), the optimal period for dredging is after reproduction either in  
702 August-September in tropical reefs or April-May in temperate reefs, when few of the major  
703 habitat-forming macroalgae are undergoing reproduction or recruitment (Figure 1). We  
704 restrict our analysis to the persistent, leathery macrophyte genera *Sargassum* and *Ecklonia* -  
705 given that they are habitat-forming macrophytes with adequate LHC data in WA.

706

#### 707 6.3.1. *Sargassum*

708 In temperate WA, the most common *Sargassum* spp. phenology is a spring-summer growth  
709 period, followed by reproduction in late summer followed by senescence, however this may  
710 not apply to tropical populations. In the temperate southwest of WA, Kendrick and Walker  
711 (1994) observed reproduction of *Sargassum* spp. during late spring - summer (September –  
712 December at Rottnest Island). Earlier, Kendrick (1993) noted that the seasonal timing of  
713 reproduction in *S. spinuligerum* varied with location and between subtidal and intertidal  
714 habitats within the same location. Patterns in reproductive phenology for tropical *Sargassum*  
715 spp. suggests that winter (July- August) is a time of senescence and low biomass after  
716 reproduction (Fulton et al., 2014). Given the variation in *Sargassum* spp. annual reproductive  
717 cycles between tropical and temperate environments, we recommend site-specific  
718 considerations when planning and managing dredging that could impact *Sargassum* spp.  
719 beds.

720

### 6.3.2. *Ecklonia*

721 Production of zoospores by the leathery macrophyte *Ecklonia radiata*. sporophytes in  
722 temperate habitats is seasonal, primarily occurring between early summer and autumn  
723 (December – May), with a peak in April (Mohring et al., 2013a; Mohring et al., 2013b).  
724 Based on the assumption that the period leading up to sporangial production, spore release  
725 and then gametophyte growth is sensitive to perturbation, we suggest that winter would be  
726 the optimal period for dredging in temperate WA. Winter is also the season of slowest  
727 growth, and significant thallus erosion and dislodgement due to storm conditions. Underlying  
728 juvenile sporophytes may also contribute to the formation of a new canopy following a  
729 canopy loss, but this is dependent on the timing of canopy removal, with late summer –  
730 autumn loss favouring faster recovery (Toohey and Kendrick, 2007).

731  
732  
733 In general, dredging during winter is likely to be the most effective EW to use for both  
734 seagrasses and macroalgae in temperate WA, given that canopy forming seagrasses and  
735 macroalgae are less sensitive to changes in light availability during this time period due to  
736 timing of LHCs, and the least number of invertebrate species are undergoing periods  
737 spawning and recruitment. However, this EW may have to be adjusted based on cumulative  
738 impacts from pre-existing anthropogenic or natural stressors that may overlap in time and  
739 space with impacts from dredging (Erfteemeijer et al, 2006). In addition, lag effects - where  
740 there is a delay between the stressor and the physiological responses - would also have to be  
741 carefully monitored for, and timing of EWs changed if required (Atkins et al, 2011).

## 742 743 **7. Management implications and future work**

744 Dredging has the potential to have adverse impacts on benthic marine organisms. EWs, or the  
745 avoidance of intensive dredging activities during ecologically sensitive periods, may  
746 sometimes be an effective management tool to prevent significant impacts. This requires  
747 location-specific knowledge of the timing of sensitive periods in the life histories of the  
748 organisms present and a contextual understanding of local environmental conditions. Where  
749 large uncertainties exist regarding the probable responses of benthos to dredging-induced  
750 increased exposures to suspended or deposited sediments, EWs represent a logical approach  
751 to dredging project management. Resort to an EW should, however, take into consideration  
752 other potentially effective dredging project management practices (PIANC 2009). Trade-offs



753 are inherent in decisions to implement any dredging management practice. For example,  
754 deployment of silt curtains to protect seagrass beds has numerous logistical and economic  
755 constraints. In the United States, which has an extensive history of applying EWs, the NRC  
756 (2002) recommended a structured approach involving coordination between engineers  
757 charged with proposing an optimal set of dredging equipment and management controls and  
758 scientists charged with identifying optimal protection strategies.

759

760 In Western Australia, local knowledge of potentially critical life history periods (such as  
761 reproduction and recruitment) is lacking for many dominant species of invertebrates,  
762 seagrasses and macroalgae. A range of marine organisms in temperate WA exhibit an  
763 increased vulnerability to disturbance during the summer months (Oct. – April) due to the  
764 timing of sensitive life history periods, which suggests that the winter months may represent  
765 a potentially effective EW for dredging in this region. Moving further north into tropical WA,  
766 the timing of vulnerability changes to autumn-winter and for some species, spring (April -  
767 Oct.) where the strong seasonality of the wet summer months enhances growth, reproductive  
768 and recruitment timing. Further north, the wet season impacts some groups more than others  
769 (Hovey et al. 2015) and EWs for seagrasses for example would be better placed over summer  
770 during the wet season (Nov. – March).

771

772 There is currently insufficient scientific basis to justify the adoption of any generic  
773 environmental windows for dredging operations in WA for benthic organisms other than  
774 corals, and even that is limited for temperate reefs. We suggest the following general criteria  
775 for evaluating the potential of using EWs to mitigate dredging impacts:-

776 1. Identify ecologically, economically, or socially ‘valuable’ species present in order to  
777 prioritise protection (Arponen 2012; Costanza et al, 2014). For example, habitat-  
778 forming seagrasses, macroalgae, or invertebrates would be classified as high priority  
779 given their ecological importance as habitat and food for other species in the  
780 ecosystem including humanity.

781 2. Identify life history characteristics of species to determine overall vulnerability to  
782 dredging impacts, with particular emphasis on priority species.

783 3. Identify potential environmental windows for priority species in cases where dredging  
784 impacts cannot be confidently minimised by implementing other management  
785 practices. For example, dredging outside of reproduction and recruitment periods for  
786 invertebrates.

787 4. Combine environmental windows for priority species with knowledge of local  
788 environmental conditions and potential thresholds for dredging impacts to determine  
789 periods when the impacts of dredging could be minimized.

790

791 These criteria must be continuously updated as new data on LHCs are obtained or as  
792 environmental conditions and species vulnerabilities change, forming part of the framework  
793 that can assess risks to guide bridging and planning for dredging. Although detailing  
794 monitoring methods is beyond the scope of this review, recognition should be given that  
795 monitoring the performance of EWs and other dredging project management practices must  
796 be conducted in order to refine and improve protection measures for future dredging projects.

797

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**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
A. 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
B. Seagrasses	Growth rate	Slow-growing, persistent		Fast-growing, colonising
	Time to sexual maturity	Long		Short
	Turnover time	Slow		Fast
	Seed bank presence	Absent		Present
C. Macroalgae	Growth rate	Slow-growing leathery macrophyte		Fast-growing turf or siphonous algae
	Lifespan	Longer-lived (years)		Shorter-lived (days–months)
	Reproductive strategy	Less complex (fewer stages)		More complex (more stages)

**Table 2.** Summary of *Posidonia* spp. responses to (A) light reduction and (B) sediment burial.

	Species	Light Level	Duration	Response	Recovery	Reference
A. Light reduction	<i>P. australis</i>	Sub-lethal	90 days	Decrease in shoot density and biomass	Little recovery	(Fitzpatrick and Kirkman, 1995)
		Sub-lethal	46 months	Decrease in shoot density	None (slow decline)	(Fyfe and Davis, 2007)
	<i>P. sinuosa</i>	0-10%SI	148 days	Decrease in shoot density and productivity	Little recovery 3.5-4 years (3-4 m depth); 5 years (7-8 m depth)	(Gordon et al., 1994) (Collier et al., 2009)
	<i>Species</i>	Burial depth	Duration	Response	Recovery	Reference
B. Burial	<i>P. augustifolia</i>		2 weeks	Total mortality		(Clarke, 1987)
	<i>P. australis</i>	>15 cm	50 days	50% mortality		(Cabaço et al., 2008)
	<i>P. oceanica</i>	5 cm		Decreased biomass	Not measured	(Erfteimeijer et al., 2006)
	<i>P. sinuosa</i>	>15 cm	50 days	50% mortality		(Cabaço et al., 2008)

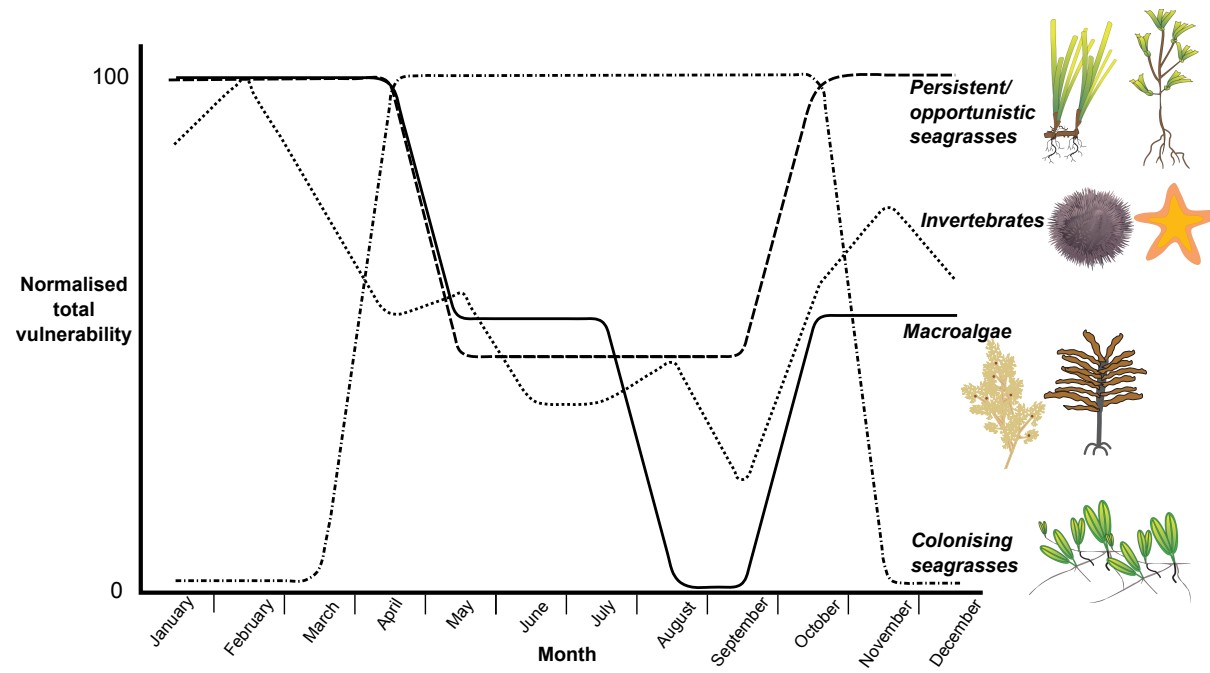






Taxa	Representative species	J	F	M	A	M	J	J	A	S	O	N	D	
	<i>Ecklonia</i> <sup>2</sup>													

<sup>1</sup>Representative species from WA; <sup>2</sup>Representative genus from WA; <sup>3</sup>Representative species elsewhere in Australia; <sup>4</sup>Representative genus elsewhere in Australia;  
<sup>5</sup>Representative species overseas; <sup>6</sup>Representative genus overseas.



**Figure 1.** Normalised total annual vulnerability based on the timing of sensitive life history periods (Table 3) and vulnerability scores (Appendices B, D & E) for representative species of invertebrates, seagrasses and macroalgae in Western Australia.