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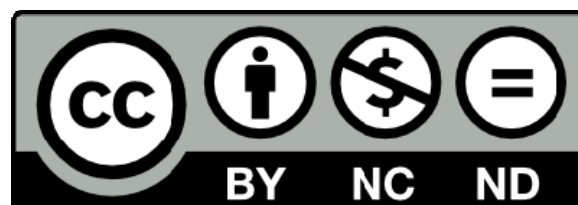
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1 Effects of dredging on critical ecological processes for marine invertebrates, seagrasses and
 2 macroalgae, and the potential for management with environmental windows using Western
 3 Australia as a case study

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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⁻¹, 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 μmol
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 *Breyenia 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

721 6.3.2. *Ecklonia*

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

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> ²													

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

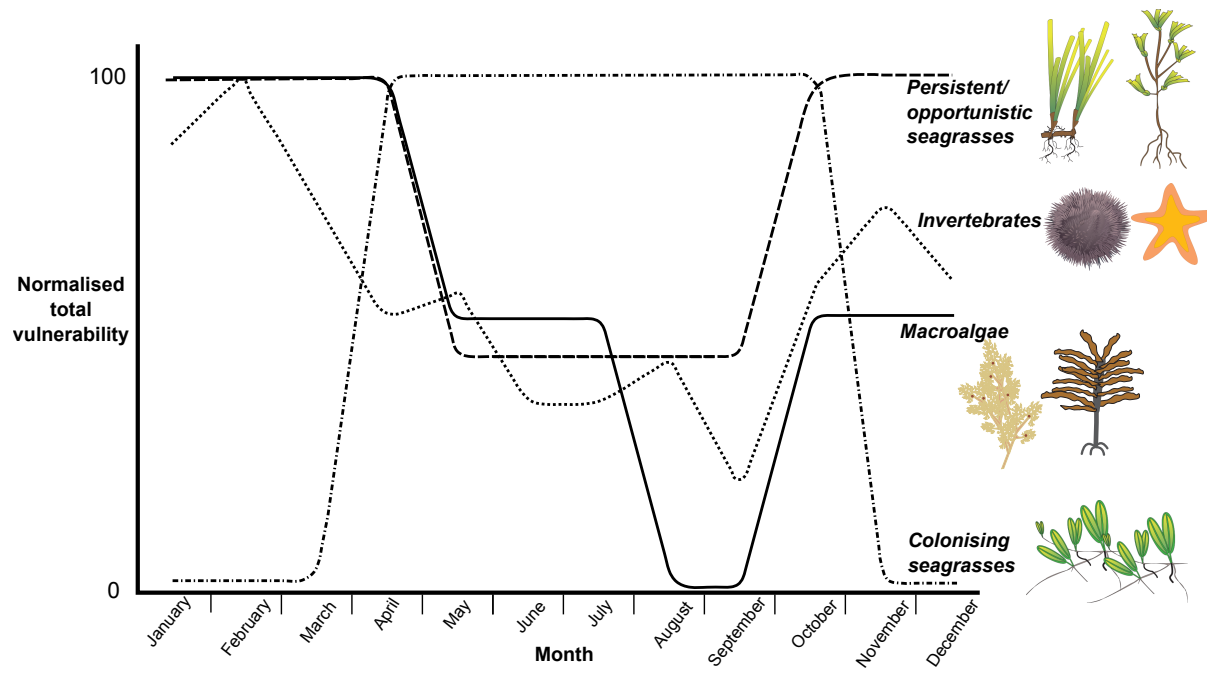


Figure 1. Normalised total annual vulnerability based on the timing of sensitive life history periods (Table 3) and vulnerability scores (Appendices B, D & E) for representative species of invertebrates, seagrasses and macroalgae in Western Australia.