2017

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

Matthew Fraser
*University of Western Australia*

Jessie Short
*University of Western Australia*

Gary A. Kendrick
*University of Western Australia*, gary.kendrick@uwa.edu.au

Dianne McLean
*University of Western Australia*

John Keesing
*Commonwealth Scientific Industrial Research Organisation*

*See next page for additional authors*

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

**Abstract**

Dredging can have significant impacts on benthic marine organisms through mechanisms such as sedimentation and reduction in light availability as a result of increased suspension of sediments. Phototrophic marine organisms and those with limited mobility are particularly at risk from the effects of dredging. The potential impacts of dredging on benthic species depend on biological processes including feeding mechanism, mobility, life history characteristics (LHCs), stage of development and environmental conditions. Environmental windows (EWs) are a management technique in which dredging activities are permitted during specific periods throughout the year; avoiding periods of increased vulnerability for particular organisms in specific locations. In this review we identify these critical ecological processes for temperate and tropical marine benthic organisms; and examine if EWs could be used to mitigate dredging impacts using Western Australia (WA) as a case study. We examined LHCs for a range of marine taxa and identified, where possible, their vulnerability to dredging. Large gaps in knowledge exist for the timing of LHCs for major species of marine invertebrates, seagrasses and macroalgae, increasing uncertainty around their vulnerability to an increase in suspended sediments or light attenuation. We conclude that there is currently insufficient scientific basis to justify the adoption of generic EWs for dredging operations in WA for any group of organisms other than corals and possibly for temperate seagrasses. This is due to; 1) the temporal and spatial variation in the timing of known critical life history stages of different species; and 2) our current level of knowledge and understanding of the critical life history stages and characteristics for most taxa and for most areas being largely inadequate to justify any meaningful EW selection. As such, we suggest that EWs are only considered on a case-by-case basis to protect ecologically or economically important species for which sufficient location-specific information is available, with consideration of probable exposures associated with a given mode of dredging.

**Authors**
Matthew Fraser, Jessie Short, Gary A. Kendrick, Dianne McLean, John Keesing, Maria Byrne, M Julian Caley, Doug Clarke, Andrew R. Davis, Paul Erftemeijer, Stuart Field, Sam Gustin-Craig, John M. Huisman, Michael J. Keough, Paul Lavery, Ray Masini, Kathryn McMahon, Kerrie Mengersen, Michael Rasheed, John Statton, Jim Stoddart, and Paul Wu

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
4

Matthew W. Fraser[1,2*†], Jessie Short[1,2†], Gary Kendrick[1,2], Dianne McLean[1,2], John Keesing[1,8], Maria Byrne[3], M. Julian Caley[4], Doug Clarke[5], Andrew R. Davis[6], Paul L.A. Erftemeijer[2], Stuart Field[2,14], Sam Gustin-Craig[1,2], John Huisman[7,15], Mick Keough[9], Paul S. Lavery[1,10], Ray Masini[1,11], Kathryn McMahon[1,10], Kerrie Mengersen[12], Michael Rasheed[13], John Statton[1,2] , Jim Stoddart[2], Paul Wu[12]

5

[1]Western Australian Marine Science Institution, Perth, WA 6000, Australia

[2]The UWA Oceans Institute and School of Plant Biology, The University of Western Australia, 35 Stirling Highway, Crawley, Western Australia 6009, Australia.

[3]Schools of Medical and Biological Sciences, University of Sydney, Sydney, NSW 2006, Australia

[4]Australian Institute of Marine Science, Townsville, QLD 4810, Australia

[5]U.S. Army Engineer Research and Development Center (CEERD-EM-D), Vicksburg, MS 39180-6199, U.S.A.

[6]Centre for Sustainable Ecosystem Solutions and School of Biological Sciences, University of Wollongong, Wollongong, NSW 2522, Australia

[7]School of Veterinary and Life Sciences, Murdoch University, Murdoch, WA 6150, Australia

[8]CSIRO Marine and Atmospheric Research, Crawley, WA 6009 Australia

[9]Department of Zoology, Centre for Aquatic Pollution Identification and Management (CAPIM), The University of Melbourne, VIC 3010, Australia

[10]School of Science and Centre for Marine Ecosystems Research, Edith Cowan University, Perth, WA 6050, Australia

[11]Office of the Environmental Protection Authority, Perth, Western Australia 6000, Australia

[12]School of Mathematical Sciences, Queensland University of Technology, Brisbane, QLD 4001, Australia

[13]Centre for Tropical Water and Aquatic Ecosystem Research, James Cook University, Cairns and Townsville, QLD 4870, Australia

[14] Department of Parks and Wildlife, Kensington, WA 6151, Australia

[15] Western Australian Herbarium, Department of Parks and Wildlife, Locked Bag 104, Bentley Delivery Centre, WA 6983

*Corresponding author: matthew.fraser@uwa.edu.au

[†]These authors contributed equally to this work.

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

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

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

Highlights:

- Reducing dredging during sensitive life history periods may reduce dredging impacts.

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

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

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

## 1. Introduction

44

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

*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

### 3.1.3. Reproductive and developmental strategy

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

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

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

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

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

### 4.2.2. Opportunistic seagrass species

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

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

432     2015), unless the seeds are buried under too much sediment, preventing the hypocotyl from

433     penetrating the sediment surface (Birch, 1981). In addition, there is a trade-off between fast

434     growth and reproduction, which results in a relatively low tolerance to prolonged periods of

435     decreased light levels compared to more persistent species. *Halophila* species are able to

436     physiologically and morphologically acclimate to reductions in available light due to their

437     relatively small size. *Halophila ovalis* shows acclimation potential to light levels below their

438     minimum light requirements, but only for 3-5 days, after which growth rates are reduced

439     (Longstaff et al., 1999a,b). Recovery was possible for this species if light levels were restored

440     within 9 days, but periods of low light exceeding 15 days were associated with an

441     exponentially greater risk of mortality, with 100% mortality occurring after 30 days of

442     shading (Longstaff et al., 1999a). Acclimation to chronic low light conditions in *Halophila*

443     *ovalis* in Singapore waters was found to reduce its resilience to further (short-term) sediment

444     disturbances (Yaakub et al., 2014). Similarly, the capacity for recovery after a loss is high in

445     *Halodule wrightii*, with documented recovery of 2000 ha of seagrass in Tampa Bay, Florida,

446     USA, due to improved water conditions (Johansson, 2002).

447

448         4.2.4.   Implications for selecting environmental windows for seagrasses

449     The use of EWs prior to dredging can be important for ensuring seagrass recovery following

450     a dredging event. EWs must take seagrass biology, phenology and environmental seasonality

451     for each species at each site into account. Life span, growth rate and reproductive strategy are

452     critical aspects of plant biology that contribute to the overall vulnerability of seagrass species

453     (Table 1(B)), while important phenological considerations include flowering, fruiting and

454     sensitive life-history stages (e.g. seedling vs. mature plant stages). Furthermore, the most

455     sensitive period in the life cycle may depend on the life-history strategy of a particular

456     species. For example, for persistent species the adult plant would take priority for protection,

457     whereas for colonising species, periods of reproduction and production of seedbanks would

458     be more vulnerable and take priority. A summary of characteristics that contribute to seagrass

459     vulnerability to dredging is given in Table 1(B). In general, slow-growing seagrasses that

460     take longer to reach sexual maturity and do not form sediment seed banks will have a higher

461     vulnerability than fast-growing seagrasses with short turnover times, shorter periods to reach

462     sexual maturity, and sedimentary seed banks. In addition, seasonal environmental cycles such

463     as light, temperature, tidal cycles (i.e. periods of exposure), storms and other periods of high

464     swell/wave energy must be considered. Assessing the temporal cycle of plant phenology

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, S*argassum* 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

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

### 5.1.3. Crustose Coralline Algae

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

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

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 *Breynia 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 that 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 (Erftemeijer 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

806
807
808

References

Aldridge, D.W., Payne, B.S., Miller, A.C., 1987. The effects of intermittent exposure to suspended solids and turbulence on three species of freshwater mussels. Environmental Pollution 45, 17-28.

Ang, P.O., 1985. Phenology of Sargassum siliquosum J. Ag and S. paniculatum J. Ag. (Sargassaceae, Phaeophyta) in the reef flat of Balibago (Calatagan, Philippines), 5th International Coral Reef Congress, pp. 51-57.

Ang, P.O., 2007. Phenology of Sargassum spp. in Tung Ping Chau Marine Park, Hong Kong SAR, China, International Seaweed Symposium, pp. 403-410

Babcock, R., Wills, B., Simpson, C., 1994. Mass spawning of corals on a high latitude coral reef. Coral Reefs 13, 161-169.

Arponen, A., 2012. Prioritizing species for conservation planning. Biodiversity and Conservation 4, 875-893.

Atkins, J.P., Burdon, D., Elliott, M., Gregory, A.J., 2011. Management of the marine environment: Integrating ecosystem services and societal benefits with the DPSIR frameworkin a systems approach. Marine Pollution Bulletin 62, 215-226.

Babcock, R.C., Bull, G.D., Harrison, P.L., Heyward, A.J., Oliver, J.K., Wallace, C.C., Willis, B.L., 1986. Synchronous spawnings of 105 scleractinian coral species on the Great Barrier Reef. Marine Biology 90, 379-384.

Babcock, R.C., Mundy. C., Keesing, J., Oliver, J., 1992. Predictable and unpredictable spawning events: in situ behavioural data from free-spawning coral reef invertebrates. Invertebrate Reproduction & Development 22, 213-227.

Baird, A.H., Blakeway, D.R., Hurley, T.J., Stoddart, J.A., 2011. Seasonality of coral reproduction in the Dampier Archipelago, northern Western Australia. Marine Biology 158, 275-285.

Bell, J.J., McGrath, E., Biggerstaff, A., Bates, T., Bennett, H., Marlow, J., Shaffer, M., 2015. Sediment impacts on marine sponges. Marine Pollution Bulletin 94, 5-13.

Bender, K., Jensen, K., 1992. The effect of the construction works at the Great Belt link on mussel (Mytilus edulis) beds, Proceedings of the 12th Baltic Marine Biologists Symposium: Physiological and Biochemical Strategies in Baltic Organisms; New Approaches in Ecotoxicological Research; Changes in Baltic Ecosystems, Especially Those Induced by Engineering Activities, Helsingør, Denmark, 25-30 August 1991. Olsen & Olsen, pp. 17.

Birch, W.R., 1981. Morphology of germinating seeds of the seagrass Halophila spinulosa (R.Br.) Achers. (Hydrocharitaceae). Aquatic Botany 11, 79-90.

Bulthuis, D.A., 1983. Effects of *in situ* light reduction on density and growth of the seagrass Heterozostera tasmanica (Martens ex Aschers.) den Hartog in Western Port, Victoria, Australia. Journal of Experimental Marine Biology and Ecology 67, 91-103.

Bulthuis, D.A., Woelkerling, W.J., 1983. Seasonal variation in standing crop, density and leaf growth rate of the seagrass, Heterozostera tasmanica in western Port and Port Phillip Bay, Victoria, Australia. Aquatic Botany 16, 111-136.

Cabaço, S., Santos, R., 2007 Effects of burial and erosion on the seagrass Zostera nolti. Journal of Experimental Marine Biology and Ecology 340, 204 -212.

Cabaço, S., Santos, R., Duarte, C.M., 2008. The impact of sediment burial and erosion on seagrasses: a review. Estuarine, Coastal and Shelf Science 79, 354-366.

Cambridge, M.L. 1975 Seagrasses of south-western Australia with special reference to the ecology of Posidonia australis Hook f. in a polluted environment. Aquatic Botany 1, 149-161.

Campbell, S.J., McKenzie, L.J., 2004. Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. Estuarine, Coastal and Shelf Science 60, 477-490.

Campey, M.L., Kendrick, G.A., Walker, D.I., 2002. Interannual and small-scale spatial variability in sexual reproduction of the seagrasses Posidonia coriacea and Heterozostera tasmanica, southwestern Australia. Aquatic botany 74, 287-297.

Caputi, N., Penn, J.W., Joll, L.M., Chubb, C.F., 1998. Stock-recruitment-environment relationships for invertebrate species of Western Australia. Canadian Special Publication of Fisheries and Aquatic Sciences, 247-256.

Carballo, J.L., 2006. Effect of natural sedimentation on the structure of tropical rocky sponge assemblages. Ecoscience 13, 119-130.

Chapman, A.S., Fletcher, R.L., 2002. Differential effects of sediments on survival and growth of Fucus serratus embryos (Fucales, Phaeophyceae). Journal of Phycology 38, 894-903.

Chartrand KM, Bryant CV, Carter AB, Ralph PJ and Rasheed MA (2016) Light Thresholds to Prevent Dredging Impacts on the Great Barrier Reef Seagrass, Zostera muelleri ssp. capricorni. Frontiers of Marine Science 3:106. doi: 10.3389/fmars.2016.00106

Chevron, 2009. Gorgon gas development and Jansz feed gas pipeline: dredging and dredge spoil management and monitoring plan. Document No. G1-NT-PLNX0000373. Chevron Australia: Perth.

Chew, C.A., Hepburn, C.D., Stephenson, W., 2013. Low-level sedimentation modifies behaviour in juvenile Haliotis iris and may affect their vulnerability to predation. Marine Biology 160, 1213-1221.

Chittleborough, R.G., Thomas, L.R., 1969. Larval ecology of the Western Australian marine crayfish, with notes upon panulirid larvae from the eastern Indian Ocean. Marine and Freshwater Research 20, 199-224.

Clarke, S., 1987. Seagrass-sediment dynamics in Holdfast Bay: summary. Safish 11, 4-10.

Clarke, S.M., Kirkman, H., 1989. Seagrass dynamics. Elsevier, Netherlands, 304- 345 pp.

Clifton, K.E., Clifton, L.M., 1999. The phenology of sexual reproduction by green algae (Bryopsidales) on Caribbean coral reefs. Journal of Phycology 35, 24-34.

Collier, C.J., Lavery, P.S., Masini, R.J., Ralph, P.J., 2007. Morphological, growth and meadow characteristics of the seagrass Posidonia sinuosa along a depth-related gradient of light availability. Marine Ecology Progress Series 337, 103-115.

Collier, C.J., Lavery, P.S., Ralph, P.J., Masini, R.J., 2009. Shade-induced response and recovery of the seagrass Posidonia sinuosa. Journal of Experimental Marine Biology and Ecology 370, 89-103.

Cortés, J.N., Risk, M.J., 1985. A reef under siltation stress: Cahuita, Costa Rica. Bulletin of Marine Science 36, 339-356.

Costanza R., de Groot R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Global Environmental Change 26, 152-158.

Crane, K.R., 1999. Reproductive biology of Scleractinian coral at Rottnest Island, Western Australia. Honours Thesis (160 pp.), Murdoch University, Western Australia.

Dawes, C.J., Andorfer, J., Rose, C., Uranowski, C., Ehringer, N., 1997. Regrowth of the seagrass Thalassia testudinum into propeller scars. Aquatic Botany 59, 139-155.

de Bettignies T., Wernberg, T., Lavery, P.S., Vanderklift, M.A., Mohring, M.B. 2013. Mechanisms of dislodgement and erosion contribute to production of kelp detritus. Limnology & Oceanography 58: 1680-1688.

De Lange H.J., Sala, S., Vighi M., Faber, J.H., 2010. Ecological vulnerability in risk assessment - A review and perspectives. Science of the Total Environment 408, 3871-3879.

De Wreede, R.E., 1976. The phenology of three species of Sargassum (Sargassaceae, Phaeophyta) in Hawaii*. Phycologia 15, 175-183.

Demers, M.A., Davis, A.R., Knott, N.A., 2013. A comparison of the impact of 'seagrass-friendly' boat mooring systems on Posidonia australis. Marine Environmental Research 83, 54-62.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P.W., Batiuk, R.A., 1993. Assessing water quality with submersed aquatic vegetation. BioScience 43, 86-94.

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. ICES Journal of Marine Science: Journal du Conseil 57, 1428-1438.

Duarte, C.M., 1991. Seagrass depth limits. Aquatic Botany 40, 363-377.

Duarte, C.M., Terrados, J., Agawin, N.S., Fortes, M.D., Bach, S., Kenworthy, W.J., 1997. Response of a mixed Philippine seagrass meadow to experimental burial. Marine Ecology Progress Series 147, 285-294.

EPA, 2011. Environmental Assessment Guidline for Marine Dredging Proposals. In: Environmental Protection Authority, Environmental Assessment Guidelines 7, Western Australia.

Erftemeijer, P.L.A., Lewis, R.R. III, 2006. Environmental impacts of dredging on seagrasses: A review. Marine Pollution Bulletin 52, 1553-1572.

Erftemeijer, P.L.A., Riegl, B., Hoeksema, B.W., Todd, P.A., 2012. Environmental impacts of dredging and other sediment disturbances on corals: a review. Marine Pollution Bulletin 64, 1737-1765.

Essink, K., 1999. Ecological effects of dumping of dredged sediments; options for management. Journal of Coastal Conservation 5, 69-80.

Fabricius, K., De'Ath, G., 2001. Environmental factors associated with the spatial distribution of crustose coralline algae on the Great Barrier Reef. Coral Reefs 19, 303-309.

Filho, G.M.A., Creed, J.C., Adrade, L.R., Pfeiffer, W.C. 2004. Metal accumulation by Halodule wrightii populations. Aquatic Botany 80, 241-251.

Fitzpatrick, J., Kirkman, H., 1995. Effects of prolonged shading stress on growth and survival of seagrass Posidonia australis in Jervis Bay, New South Wales, Australia. Marine Ecology Progress Series 127, 279-289.

Fulton, C.J., Depczynski, M., Holmes, T.H., Noble, M.M., Radford, B., Werenberg, T., Wilson, S.K., 2014. Sea temperature shapes seasonal fluctuations in seaweed biomass within the Ningaloo coral reef ecosystem. Limnology and Oceanography 59, 156-166.

Garrido, M., Lafabrie, C., Torre, F., Fernandez, C., Pasqualini, V., 2013. Resilience and stability of Cymodocea nodosa seagrass meadows over the last four decades in a Mediterranean lagoon. Estuarine, Coastal and Shelf Science 130, 89-98.

Gerrodette, T., Flechsig, A.O., 1979. Sediment-induced reduction in the pumping rate of the tropical sponge Verongia lacunosa. Marine Biology 55, 103-110.

Herrnkind, W.F., Butler Iv, M.J., Tankersley, R.A., 1988. The effects of siltation on recruitment of spiny lobsters, Panulirus argus. Fishery Bulletin 86, 331-338.

Hillis-Colinvaux, L., 1986. Deep water populations of Halimeda in the economy of an atoll. Bulletin of Marine Science 38, 155-169.

Hovey, R.K., Cambridge, M.L., Kendrick, G.A., 2012. Season and sediment nutrient additions affect root architecture in the temperate seagrasses Posidonia australis and P. sinuosa. Marine Ecology Progress Series 446, 23-30.

Hovey, R.K., Zavala Perez, A., Statton, J., Fraser, M.W., Ruiz Montoya, L., Rees, M., Stoddart, J., Kendrick, G.A. 2015. Strategy for assessing impacts in highly seasonal tropical seagrasses. Marine Pollution Bulletin 101, 594-599.

Johansson, J., 2002. Historical overview of Tampa Bay water quality and seagrass issues and trends. Seagrass Management: It's Not Just Nutrients! Symposium Proceedings, St. Petersburg, Florida, USA.

Joll, L.M., 1980. Reproductive biology of two species of Turbinidae (Mollusca: Gastropoda). Marine and Freshwater Research 31, 319-336.

Jones, R., Ricardo, G.F., Negri, A.P., 2015. Effects of sediments on the reproductive cycle of corals. Marine Pollution Bulletin 100, 13-33.

Keats, D., Knight, M., Pueschel, C., 1997. Antifouling effects of epithallial shedding in three crustose coralline algae (Rhodophyta, Coralinales) on a coral reef. Journal of Experimental Marine Biology and Ecology 213, 281-293.

Keesing, J.K, Graham F., Irvine, T. and Crossing, R., 2011. Synchronous aggregated pseudocopulation of the sea star Archaster angulatus Muller and Troschel, 1842 (Echinodermata: Asteroidea) and its reproductive cycle in south-western Australia. Marine Biology 158, 1163–1173

Keesing, J.K., Usher, K.M., Fromont, J., 2012. First record of photosynthetic cyanobacterial symbionts from mesophotic temperate sponges. Marine and Freshwater Research 63, 403-408.

Keesing J.K., Irvine, T.R., 2013. Aspects of the biology of an abundant spatangoid urchin, Breynia desori in the Kimberley region of north-western Australia. p165-174, in: Johnson, C.R. (ed.) Echinoderms in a Changing World. CRC Press/Balkema, The Netherlands.

Kendrick, G.A., 1993. Sargassum beds at Rottnest Island: Species composition and abundance. In: Wells, F., Walker, D., Kirkman, H., Lethbridge, R. (Eds.), Proceedings of the 5th International Marine Biology Workshop: The Marine Flora and Fauna of Rottnest Island, Western Australia, Western Australian Museum, Perth, pp. 455–472.

Kendrick, G.A., Walker, D.I., 1994. Role of recruitment in structuring beds of *Sargassum* spp. (Phaeophyta) at Rottnest Island, Western Australia Journal of Phycology 30, 200-208.

Kilminster, K., McMahon, K., Waycott, M., Kendrick, G.A., Scanes, P., McKenzie, L., O'Brien, K.R., Lyons, M., Furguson, A., Maxwell, P., Glasby, T., Udy, J., 2015. Unravelling complexity in seagrass systems for management: Australia as a microcosm. Science of the Total Environment 534,97-109.

Kirkman, H., 1978. Decline of seagrass in northern areas of Moreton Bay, Queensland. Aquatic Botany 5, 63-76.

Kirkman, H., 1999. Pilot experiments on planting seedlings and small seagrass propagules in Western Australia. Marine Pollution Bulletin 37, 460-467.

Kiswara, W., Huiskes, A.H.L., Herman, P.M.J., 2005. Uptake and allocation of $^{13}$C by Enhalus acoroides at sites differing in light availability. Aquatic Botany 81, 353 - 366.

Lanyon, J.M., Marsh, H., 1995. Temporal changes in the abundance of some tropical intertidal seagrasses in North Queensland. Aquatic Botany 49, 217-237.

Lavery, P.S., McMahon, K., Mulligan, M., Tennyson, A., 2009. Interactive effects of timing, intensity and duration of experimental shading on Amphibolis griffithii. Marine Ecology Progress Series 394,21-33.

Lee, K.S., Dunton, K.H., 1997. Effect of in situ light reduction on the maintenance, growth and partitioning of carbon resources in Thalassia testudinum banks ex König. Journal of Experimental Marine Biology and Ecology 210, 53-73.

Lemloh, M.-L., Fromont, J., Brümmer, F., Usher, K.M., 2009. Diversity and abundance of photosynthetic sponges in temperate Western Australia. BMC Ecology 9, 4.

Linstone, H.A., Turoff, M. (Eds.) 1975. The Delphi method: Techniques and applications (Vol. 29). Reading, MA: Addison-Wesley.

Longstaff, B., Loneragan, N., O'donohue, M., Dennison, W., 1999a. Effects of light deprivation on the survival and recovery of the seagrass Halophila ovalis (R. Br.) Hook. Journal of experimental marine biology and ecology 234, 1-27.

Longstaff, B.J., Dennison, W.C., 1999b. Seagrass survival during pulsed turbidity events: the effects of light deprivation on the seagrasses Halodule pinifolia and Halophila ovalis. Aquatic Botany 65, 105-121.

Maggs, C.A., Guiry, M.D., 1987. Environmental control of macroalgal phenology. Blackwell Scientific Publications, Oxford, U.K.

Marba, N., Duarte, C.M., 1994. Growth response of the seagrass Cymodocea nodosa to experimental burial and erosion. Marine Ecology Progress Series. 107(3), 307-311.

Marba, N., Hemminga, M.A., Duarte, C.M., 2006. Resource translocation within seagrass clones: allometric scaling to plant size and productivity. Oecologia 150, 362 - 372.

Marszalek, D.S., 1981. Impact of dredging on a subtropical reef community, southeast Florida, USA, 4. International Coral Reef Symposium, Manila (Philippines), 18-22 May 1981.

McLeod, K., Leslie, H., 2009. Ecosystem-based management for the oceans. Cambridge Univ Press.

McMahon, K., Lavery, P.S., Mulligan, M., 2011. Recovery from the impact of light reduction on the seagrass Amphibolis griffithii, insights for dredging management. Marine Pollution Bulletin 62, 270-283.

Mills, K.E., Fonseca, M.S., 2003. Mortality and productivity of eelgrass Zostera marina under conditions of experimental burial with two sediment types. Marine Ecology Progress Series 255, 127 - 134.

Mohring, M.B., Kendrick, G.A., Wernberg, T., Rule, M.J., Vanderklift, M.A., 2013a. Environmental influences on kelp performance across the reproductive period: an ecological trade-off between gametophyte survival and growth? PLoS One 8, e65310.

Mohring, M.B., Wernberg, T., Kendrick, G.A., Rule, M.J., 2013b. Reproductive synchrony in a habitat-forming kelp and its relationship with environmental conditions. Marine Biology 160, 119-126.

Mohring, M.B., Wernberg, T., Wright, J.T., Connell, S.D., Russell, B.D. 2014. Biogeographic variation in temperature drives performance of kelp gametophytes during warming. Marine Ecology Progress Series 513:85-96.

Nelson, K. 2009. Calcified macroalgae - critical to coastal ecosystems and vulnerable to change: a review. Marine and Freshwater Research 60, 787-801.

Newell, R.C., Seiderer, L.J., Hitchcock, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological

resources on the sea bed. *Oceanography and Marine Biology: An Annual Review*, *36*, pp.127-178.

NRC, 2002. A Process for Setting, Managing and Montitoring Environmental Windows for Dredging Projects, National Research Council, Transportation Research Board Special Report 262, National Academy Press, Washington, DC, pp. 96.

Ooi, J.L.S., Kendrick, G.A., Neil, K.P.V., 2011. Effects of sediment burial on tropical ruderal seagrasses are moderated by clonal integration. Continental Shelf Research 31, 1945 -1954.

Orth, R.J., Carruthers, T.J., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., 2006. A global crisis for seagrass ecosystems. BioScience 56, 987-996.

Penn, J.W., 1980. Spawning and fecundity of the western king prawn, Penaues latisulcatus Kishinouye in western Australian waters. Marine and Freshwater Research 31, 21-35.

PIANC, 2009. Dredging Management Practices for the Environment: A Structured Selection Approach, Environmental Commission Report 100, The World Association for Waterborne Transport Infrastructure (PIANC), pp. 62.

PIANC, 2010. Dredging and Port Construction around Coral Reefs, Environmental Commission Report 108, The World Association of Waterborne Transport Infrastructure (PIANC), pp. 75.

Piazzi, L., Balata, D., Ceccherelli, G., Cinelli, F., 2005. Interactive effect of sedimentation and Caulerpa racemosa var. cylindracea invasion on macroalgal assemblages in the Mediterranean Sea. Estuarine, Coastal and Shelf Science 64(2), 467-474.

Pineda, M.C., Duckworth, A., Webster, N., 2016. Appearance matters: sedimentation effects on different sponge morphologies. Journal of the Marine Biological Association of the United Kingdom 96, 481–492.

Powilleit, M., Graf, G., Kleine, J., Riethmüller, R., Stockmann, K., Wetzel, M., Koop, J., 2009. Experiments on the survival of six brackish macro-invertebrates from the Baltic Sea after dredged spoil coverage and its implications for the field. Journal of Marine Systems 75, 441-451.

Price, I., 1989. Seaweed phenology in a tropical Australian locality (Townsville, North Queensland). Botanica Marina 32, 399-406.

Pritchard, D.W., Hurd, C.L., Beardall, J., Hepburn, C.D., 2013. Survival in low light: photosynthesis and growth of a red alga in relation to measured in situ irradiance. Journal of Phycology 49, 867-879.

Rasheed, M.A., 1999. Recovery of experimentally created gaps within a tropical Zostera capricorni (Aschers.) seagrass meadow, Queensland Australia. Journal of Experimental Marine Biology and Ecology 235, 183-200.

Reine, K.J., Dickerson, D.D., Clarke, D.G., 1998. Environmental windows associated with dredging operations. Dredging Operations and Environmental Research Program Technical Notes Collection (TN DOER-E2), U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi, USA.

Richards, Z.T., Garcia, R.A., Wallace, C.C., Rosser, N.L., Muir, P.R., 2015. A duverse assemblage of reef corals thriving in a dynamic intertidal reef setting (Bonaparte Archipelago, Kimberley, Australia). PLoS One 10, e0117791.

Riul, P., Targino, C.H., Farias, J.D.N., Visscher, P.T., Horta, P.A., 2008. Decrease in Lithothamnion sp.(Rhodophyta) primary production due to the deposition of a thin sediment layer. Journal of the Marine Biological Association of the UK 88, 17-19.

Roberts, D.E., Davis, A.R., Cummins, S.P., 2006. Experimental manipulation of shade, silt, nutrients and salinity on the temperate reef sponge Cymbastela concentrica. Marine Ecology Progress Series, 307, 143-154.

Roberts, C.M., Hawkins, J.P., 1999 Extinction risk in the sea. Trends in Ecology and Evolution 14, 241-246.

Rogers, C.S., 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62, 185-202.

Rosser, N.L., Baird, A.H., 2009. Multi-specific coral spawning in spring and autumn in far north-western Australia. Proceedings of the 11th international Coral Reef Symposium (Vol. 1, pp. 255-370).

Salomons, W., 1985. Sediments and water quality. Environmental Technology 6, 315-326.

Schaffelke, B., 1999. Particulate organic matter as an alternative nutrient source for tropical Sargassum species (Fucales, Phaeophyceae). Journal of Phycology 35, 1150-1157.

Schönberg, C.H.L., 2016. Happy relationships between marine sponges and sediments – a review and some observations from Australia. Journal of the Marine Biological Association of the United Kingdom 96, 493–514.

Sherk, J.A., 1972. Current status of the knowledge of the biological effects of suspended and deposited sediments in Chesapeake Bay. Chesapeake Science 13, S137-S144.

Simpson, C., 1985. Mass spawning of scleractinian corals in the Dampier Archipelago and the implications for management of coral reefs in Western Australia. West Aust Dept Consev Bulletin 244, 1-35.

Simpson, C., 1991. Mass spawning of corals on Western Australian reefs and comparisons with the Great Barrier Reef. Journal of the Royal Society of Western Australia 74, 85-91.

Simpson, C., Cary, J.L., Masini, R.J., 1993. Destruction of corals and other reef animals by coral spawn slicks on Ningaloo Reef, Western Australia. Coral Reefs 12, 185-191.

Statton, J., Cambridge, M.L., Dixon, K.W., Kendrick, G.A., 2013. Aquaculture of Posidonia australis seedlings for seagrass restoration programs: effect of sediment type and organic enrichment on growth. Restoration Ecology 21, 250-259.

Steneck, R.S., Graham, M.L., Bourque, B.J., Corbett. D., Erlandson, J.M., Estes, J.A., Tegner, M.J., 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. Environmental Conservation 29, 436-459.

Stoddart, J.A., Black, R., 1985. Cycles of gametogenesis and planulation in the coral Pocillopora damicornis. Marine Ecology Progress Series 23, 153-164.

Stoddart, C.W., Stoddart, J.A., Blakeway, D.R., 2012. Summer spawning of Porites lutea from north-western Australia. Coral Reefs 31, 787-792.

Styan, C.A., Rosser, N.L., 2012. Is monitoring for mass spawning events in coral assemblages in north Western Australia likely to detect spawning? Marine Pollution Bulletin 64, 2523-2527.

Suchanek, T.H., 1983. Control of seagrass communities and sediment distribution by Callianassa (Crustacea, Thalassinidea) bioturbation. Journal of Marine Research 41, 281 - 298.

Suedel, B.C., Kim, J., Clarke, D.G., Linkov, I., 2008. A risk-informed decision framework for setting environmental windows for dredging projects. Science of the Total Environment 403, 1-11.

Suedel, B. C., Clarke, J. U., Wilkins, J., Lutz, C. H., and Clarke, D. G.  2014.  The effects of a simulated suspended sediment plume on eastern oyster (*Crassostrea virginica*) survival, growth, and condition.  Estuaries and Coasts, 38, 578.

Toohey, B.D., Kendrick, G.A., 2007. Survival of juvenile Ecklonia radiata sporophytes after canopy loss. Journal of Experimental Marine Biology and Ecology 349, 170 - 182.

Umar, M.J., McCook, L.J., Price, I.R., 1998. Effects of sediment deposition on the seaweed Sargassum on a fringing coral reef. Coral Reefs 17, 169-177.

Unsworth, R.K., Rasheed, M.A., Chartrand, K.M., Roelofs, A.J., 2012. Solar radiation and tidal exposure as environmental drivers of Enhalus acoroides dominated seagrass meadows. PLoS One 7, e34133.

van Senden, D., Taylor, D., Branson, P., 2013. Realtime turbidity monitoring and modelling for dredge impact assessment in Darwin Harbour. Australasian Port and Harbour Conference (14th : 2013 : Sydney, N.S.W.). Coasts and Ports 2013: 21st Australasian Coastal and Ocean Engineering Conference and the 14th Australasian Port and Harbour Conference. Barton, A.C.T.: Engineers Australia, 797-802.

Vermaat, J.E., Agawin, N., Fortes, M., Uri, J., Duarte, C., Marba, N., Enriquez, S., Van Vierssen, W., 1997. The capacity of seagrasses to survive increased turbidity and siltation: the significance of growth form and light use. Ambio 26, 499-504.

Walker, D., McComb, A., 1988. Seasonal variation in the production, biomass and nutrient status of Amphibolis antarctica (Labill.) Sonder ex Aschers. and Posidonia australis hook. f. in Shark Bay, Western Australia. Aquatic Botany 31, 259-275.

Waycott, M., Collier, C., McMahon, K., Ralph, P., McKenzie, L., Udy, J., Grech, A., 2007. Vulnerability of seagrasses in the Great Barrier Reef to climate change. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences 106, 12377-12381.

Wells, F.E, Keesing, J.K., 1987 Population characteristics of the gastropod Cantharidus pulcherrimus on interidal platforms of the Perth area of Western Australia. Journal of the Malacological Society of Australia 8, 23-35.

Wells, F.E., Kessing, J.K., 1989. Reproduction and feeding in the abalone Haliotis roei Gray. Marine and Freshwater Research 40, 189-197.

Wells, F.E., Sellers, R.J., 1987. Reproductive periodicity of the Chiton Acanthopleura hirtosa on intertidal platforms in the Perth area of Western Australia. Journal of the Malacological Society of Australia 8, 87-92.

Westphalen, G., Collings, G., Wear, R., Fernandes, M., Bryars, S., Cheshire, A., 2004. A review of seagrass loss on the Adelaide metropolitan coastline. South Australian Research and Development Institute.

Wilber, D.H., Clarke, D.G., 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21, 855-875.

Wilber, D.H., Brostoff, W., Clarke, D.G., and Ray, G.L. 2005. Sedimentation: Potential biological effects from dredging operations in estuarine and marine environments. DOER

Technical Notes Collection (TN DOER-E20). U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi.

Wilber, D. and Clarke, D. 2010, Dredging activities and the potential impacts of sediment resuspension and sedimentation on oyster reefs, Proceedings of the Western Dredging Association Technical Conference, June 6-9, 2010, San Juan, Puerto Rico, USA, pp. 61-69.

Yaakub, S.M., Chen, E., Bouma, T.J., Erftemeijer, P.L.A., Todd, P.A., 2014. Chronic light reduction reduces overall resilience to additional shading stress in the seagrass Halophila ovalis. Marine Pollution Bulletin 83, 467–474.

Yeo, S., Keesing, J.K. and van Keulen, M., 2015. Reproductive biology of the sand dollar, Peronella lesueuri (L. Agassiz, 1841) (Echinoidea: Clypeasteroida: Laganidae), in south-western Australia, Invertebrate Reproduction & Development 59, 141-154.

York, P., Carter, A., Chartrand, K., Sankey, T., Wells, L., Rasheed, M., 2015. Dynamics of a deep-water seagrass population on the Great Barrier Reef: annual occurrence and response to a major dredging program. Scientific Reports, 5, 1-9

**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 | *P. australis* | 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) |
| | *P. sinuosa* | 0-10%SI | 148 days | Decrease in shoot density and productivity | Little recovery | (Gordon et al., 1994) |
| | | | | | 3.5-4 years (3-4 m depth) ; 5 years (7-8 m depth) | (Collier et al., 2009) |

| | *Species* | Burial depth | Duration | Response | Recovery | Reference |
|---|---|---|---|---|---|---|
| B. Burial | *P. augustifolia* | | 2 weeks | Total mortality | | (Clarke, 1987) |
| | *P. australis* | >15 cm | 50 days | 50% mortality | | (Cabaço et al., 2008) |
| | *P. oceanica* | 5 cm | | Decreased biomass | Not measured | (Erftemeijer et al., 2006) |
| | *P. sinuosa* | >15 cm | 50 days | 50% mortality | | (Cabaço et al., 2008) |

**Table 3.** Summary of known vulnerable periods for representative species of Western Australian invertebrates, seagrasses and macroalgae. For the invertebrates periods of spawning and reproduction, and for the seagrass and macroalgae periods of reproduction, recruitment and growth are shown. For references see Appendices B, D and E. Species with unknown vulnerable periods are not shown. Superscript represents location of data.

| Taxa | | Representative species | J | F | M | A | M | J | J | A | S | O | N | D |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sponges | Encrusting | *Pione velans*[1] | | | | | ▰ | | | | | | | |
| | Encrusting with some autotrophs | *Chondrilla australiensis*[1] | | | ▰ | ▰ | ▰ | | | | | | | |
| | | *Lamellodysidea herbacea*[1] | | | ▰ | ▰ | ▰ | | | | | | | |
| | | *Clinona* spp.[2] | | | ▰ | ▰ | ▰ | | | | | | | |
| | | *Xestospongia testudinaria*[3] | | | | | | | | | | ▰ | ▰ | |
| Ascidians | Solitary | *Pyura dalbyi, Polycarpa* spp.[4] | | ▰ | | | | | | | | | | |
| | | *Didemnum*[6] | ▰ | ▰ | ▰ | | | | | | | | ▰ | ▰ |
| Bryozoans | | *Bugula*[4] | ▰ | ▰ | | | | | | | | | | |
| Molluscs | Gastropods (lecithotrophs/brooders) | *Notocypraea*[2] | | | | | | | | ▰ | | | | |
| | | *Zoila*[2] | ▰ | ▰ | | | | | | | | | ▰ | ▰ |
| | | *Austrocypraea*[2] | | | | | | | | ▰ | | | | |
| | Gastropods (planktotrophs/brooders) | *Nerita albicilla*[5] | | | | | | | | ▰ | ▰ | | | |
| | | *Trochus histrio*[2] | | | | ▰ | ▰ | | ▰ | ▰ | | | | |
| | | *Turbo bruneus*[4] | | | | | | ▰ | ▰ | | | | ▰ | ▰ |
| | Bivalves | *Tridacna* spp.[4] | | | | | | | | | | | ▰ | ▰ |
| | Cephalopods | *Sepia apama*[3] | | | | | | ▰ | ▰ | | | | | |
| | | *Octopus maorum*[3] | | | | | | | ▰ | ▰ | | | | |
| | | *Sepioteuthis australis*[3] | | | | | | | | | | ▰ | ▰ | |
| | Chitons | *Acanthopleura gemmata*[3] | ▰ | ▰ | ▰ | | | | | | | | ▰ | ▰ |
| Echinoderms | Asteroids (broadcast spawners/planktotrophs) | *Linckia laevigata*[3] | | | | | | | | | | ▰ | | |
| | Ophiuroids (broadcast spawners/lecithotrophs) | *Ophionereis dubia*[4] | ▰ | ▰ | | | | | | | | | | |

| Taxa | | Representative species | J | F | M | A | M | J | J | A | S | O | N | D |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Ophiuroids (broadcast spawners/planktotrophs) | *Ophionereis semoni*[4] | X | X | | | | | | | | | | |
| | | *Ophiactis maculosa*[4] | | | | | X | X | X | | | | | |
| | | *Ophiactis savignyi*[4] | | | | | | X | X | | | | | |
| | Echinoids (broadcast spawners/lecithotrophs) | *Heliocidaris erythrogramma*[3] | X | X | X | | | | | | | | | X |
| | Echinoids (broadcast spawners/planktotrophs) | *Diadema savignyi*[5] | X | X | X | | | | | | | | | X |
| | | *Echinometra mathaei*[1] | X | X | X | X | X | X | X | X | X | X | X | X |
| | Holothuroids (broadcast spawners, planktotrophs) | *Holothuria leucospilota*[5] | X | X | X | X | | | | | | | X | X |
| | | *Holothuria atra*[3] | X | X | X | | | | X | | | | | |
| | | *Stichopus chloronatus*[3] | X | X | X | | X | | | | | | | X |
| Crustaceans | Crabs | *Portunus pelagicus*[1] | X | | | | | | | | | X | X | |
| | | *Scylla serrata*[3] | | | | | | | | | | X | X | |
| | Prawns | *Penaeus semisulcatus*[1] | X | X | | | | | | X | X | X | | |
| Cnidaria | Soft corals | *Lobophytum crassum*[4] | | | | | | | | | | | X | |
| | Temperate hard corals | *Pocillopora damicornis*[1] | X | X | | | | | | | | | | |
| | | *Goniastrea australensis*[1] | | X | | | | | | | | | | |
| | Tropical hard corals | *Acropora*[2] | | | | X | X | | | | | | X | |
| | | *Monitpora*[2] | | | | X | X | | | | | | | |
| | | *Goniastrea australensis*[1] | | | | X | X | | | | | | | |
| | | *Porites lutea*[1] | | | | | | | | | | | | X |
| Seagrasses | Temperate species | *Posidonia*[3] | X | X | X | X | | | | | | | X | X |
| | | *Amphibolis*[3] | X | X | X | X | | | | | | | X | X |
| | | *Zostera*[3] | X | X | X | X | | | | | | | X | X |
| | Tropical species | *Thalassia*[3] | X | X | X | X | | | | | | X | X | X |
| | | *Enhalus*[3] | X | X | X | X | | | | | | X | X | X |
| | | *Halophila*[3] | | | | X | X | X | X | X | X | X | X | |
| Macroalgae | Phaeophyceae | *Sargassum*[2] | X | X | X | | | | | | | | X | X |

| Taxa | Representative species | J | F | M | A | M | J | J | A | S | O | N | D |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *Ecklonia*[2] | ████ | ████ | ████ | ████ | ████ | ████ | ████ | | | | | |

[1]Representative species from WA; [2]Representative genus from WA; [3]Representative species elsewhere in Australia; [4]Representative genus elsewhere in Australia; [5]Representative species overseas; [6]Representative genus overseas.
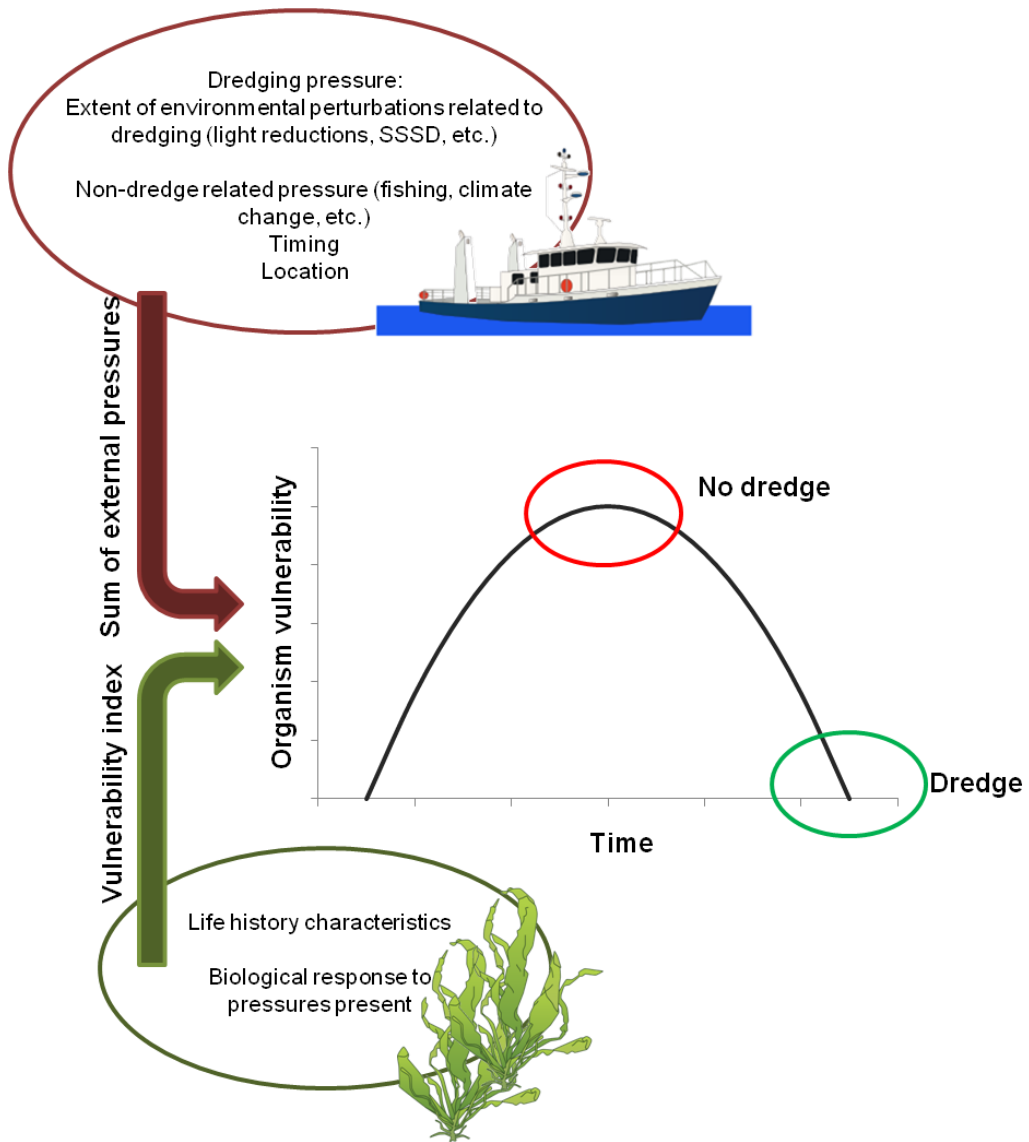
**Figure 1.** Conceptual model for the selection of effective environmental windows over an annual cycle of organism vulnerability.
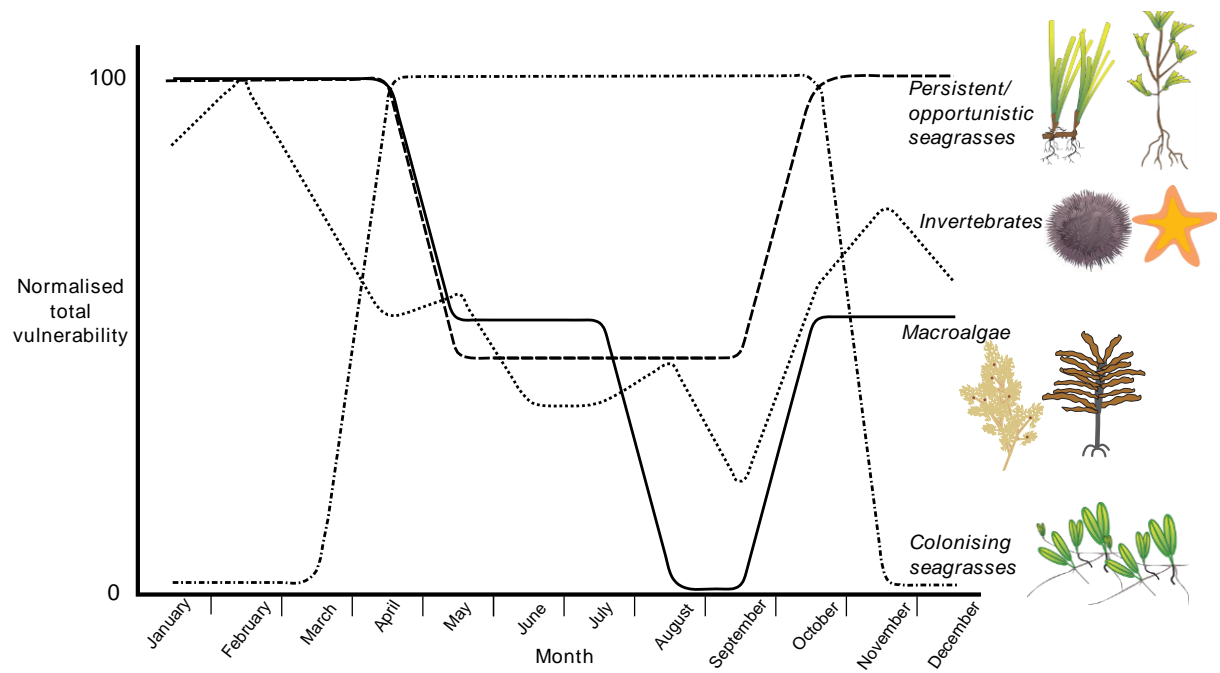
**Figure 2.** Normalised total annual vulnerability based on the timing of sensitive life history periods (Table 3) and vulnerability scores (Appendices B, D & E) for representative species of invertebrates, seagrasses and macroalgae in Western Australia.