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Improving conservation outcomes for coral reefs affected by future oil palm development in Papua New Guinea

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ABSTRACT

Clearing forests for oil palm plantations is a major threat to tropical terrestrial biodiversity, and may potentially have large impacts on downstream marine ecosystems (e.g., coral reefs). However, little is known about the impacts of runoff from oil palm plantations, so it is not clear how oil palm development should be modified to minimize the risk of degrading marine ecosystems, or how marine conservation plans should be modified to account for the impacts of oil palm development. We coupled terrestrial and marine biophysical models to simulate changes in sediment/nutrient composition on reefs as a result of oil palm development in Papua New Guinea, and predicted the response of coral and seagrass ecosystems to different land-use scenarios. The condition of almost 60% of coastal ecosystems were predicted to be substantially degraded (more than a 50% decline from their initial state) after 5 years if all suitable land was converted to oil palm, with only 4% of coastal ecosystems improving in condition as trees matured. We evaluated marine ecosystem condition if the oil palm developments were consistent with global sustainability guidelines and found that there were only slight improvements in ecosystems condition compared to the scenario with complete conversion of forest to oil palm. Substantially reducing the impact of oil palm development on marine ecosystems required limiting new plantings to hill slopes below 15°, a more stringent restriction than currently allowed for in the sustainability guidelines. We evaluated priority marine conservation areas given current land-use and found reef ecosystems in these areas will likely be heavily degraded in the future from runoff. We find that marine conservation plans should be modified to prioritize turbid areas where coral communities may be more tolerant of increased suspended sediment in the water. The approach developed here provides guidelines for modifying marine conservation priorities in areas with oil palm development. Importantly, oil palm development guidelines cannot be truly ecologically sustainable unless they are modified to account for the impacts of oil palm on coastal marine ecosystems.

1. Introduction

Coastal ecosystems are under pressure from a variety of human activities (Jackson et al., 2001). Deforestation has been shown to cause widespread destruction on the land and to downstream marine environments (Rogers, 1990). In the tropics, oil palm agriculture has been identified as a major driver of deforestation and biodiversity loss (Koh and Wilcove, 2008). The impacts of oil palm plantations to terrestrial ecosystems are clear (Fitzherbert et al., 2008), but their effects on marine ecosystems are not well understood. Erosion from new plantations can result in poor water quality from increased sediments, nutrients and pollutants (e.g., agrochemicals) (Ah Tung et al., 2009; Comte et al., 2012). To exacerbate this issue, development of palm oil plantations is occurring upstream of sensitive and biodiverse habitats, such as coral reefs.

The palm oil industry is economically important to many developing nations (Cramb and Curry, 2012), thus solutions that balance the economic benefits of oil palm with its ecological impacts are required. Poor understanding of land-sea linkages, in addition to limited data in affected regions, makes agricultural development and conservation difficult. Coral reefs are vulnerable to increases in runoff that can result from extensive land-use change, due to smothering, light loss from turbidity, eutrophication, and toxicity (Bartley et al., 2014; Fabricius, 2005; Fabricius, 2011). Despite this, the potential impact of runoff from oil palm on these ecosystems is rarely, if ever, explicitly considered during planning processes. Ignoring cross-system interactions at the land-sea interface can hinder effective conservation decisions, and may result

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in suboptimal or perverse outcomes (Álvarez-Romero et al., 2015a). Although guidelines for sustainable oil palm certification have been developed (OPIC, see http://www.rspo.org), the extent these guidelines mitigate the risks to marine biodiversity from increased runoff associated with new plantations is unknown, as cross-system impacts are not explicitly considered in the criteria for sustainability assessment.

Robust decision-making frameworks for data-poor regions that can account for land-use change, predict changes to downstream ecosystems, and identify priorities for management action are urgently needed. There are increasing numbers of approaches to modeling runoff (e.g. N-SPECT: Eslinger et al., 2005; InVEST: Tallis et al., 2013; Sednet: Wilkinson et al., 2004), but rarely do these extend into the sea. Recent studies linking runoff loads to reefs use over-simplified erosion, transport and condition models (Klein et al., 2012), or ignore the spatially and temporally heterogeneous response of different reef ecosystems to changing runoff regimes (Rude et al., 2015). Further, existing efforts to prioritize areas for marine conservation typically use only threat maps, which do not account for the greater tolerance of some ecosystems to threats than others (Tulloch et al., 2015). Importantly, no one has linked reef ecosystem condition to fine-scale land-uses and impacts in a single framework for spatial prioritization for data-limited regions.

Here we create an integrated planning framework that links land-use change under differing scenarios for the extent of oil palm expansion to their impacts on marine ecosystems in the data-limited province of New Ireland, Papua New Guinea. Our framework couples outputs from a terrestrial runoff model, ocean transport model, and ecological condition model, allowing the identification of coastal areas affected by land-use changes. Our approach builds on models of fine-scale regional erosion and coastal transport to predict sediment loads in coastal waters for data-limited regions (Álvarez-Romero et al., 2015b; Rude et al., 2015) by linking sediment loads to marine habitat condition and also accounting for changes in nutrients. We account for heterogeneity in the response of different reef ecosystems across space and time to changes in sediment and nutrient loads. Finally, we use model outputs in a marine spatial conservation prioritization that account for ecosystem condition changes from land-use changes over time. We answer the following questions:

1. How and where do changes in oil palm coverage (including using the global sustainability development guidelines) impact nutrient and sediment discharge and affect reef and seagrass ecosystems over time?
2. How do we plan for marine reserves to account for the likely impacts from expanding oil palm on the land?
3. Does incorporating oil palm development in the planning of marine reserves lead to better condition of marine ecosystems?

2. Methods

2.1. Study area

We chose a case study of the island province of New Ireland in Papua New Guinea where major current threats to marine ecosystems are fishing pressure and logging (and associated runoff), along with potential new threats from oil palm expansion (Nelson et al., 2014). We choose this region because tropical rainforests across Papua New Guinea have undergone high rates of logging and conversion to oil palm plantations in recent decades, and in island provinces, 45% of all rainforest has been logged (Shearman et al., 2009). In New Ireland Province (7404 km^2), oil palm plantations have been established at a relatively small-scale since 1994 (Koczberski et al., 2001). The province receives high levels of annual rainfall (>4500 mm), and is bordered by the Bismarck Sea in the west and the Pacific Ocean in the east, with a narrow (100 m wide), fringing reef extending down much of the northeastern coastline that drops very steeply to depths exceeding 500 m (Fig. 1).

2.1.1. Land-sea model

We applied scenarios for oil palm development to a model that coupled terrestrial processes of soil and nutrient loss with the marine processes distributing sediment and nutrients in adjacent coastal waters (Fig. 2). The models were designed for a data-limited setting with simple marine and terrestrial processes, because most oil palm development is in countries where direct measurements of runoff dispersion and habitats are not available. We predicted coral reef and seagrass condition as influenced by the indirect impacts of watershed-based pollution and direct impacts of fishing. Finally, we used the inverse of predicted ecosystem condition as the probability of degradation to plan for marine reserves that minimize the risk of destruction from different oil palm coverage scenarios, targeting high quality ecosystems for protection across the coast of New Ireland. Inputs and outputs for the model were processed using a combination of ArcGIS 10.1 and the R programming language (R Core Team, 2014) (Fig. 2). Further details for each step are described below.

2.1.2. Step 1: terrestrial runoff model

We used the open-source version of the runoff simulation tool N-SPECT (Nonpoint Source Pollution and Erosion Comparison Tool) (Eslinger et al., 2005) in MapWindow GIS to simulate runoff and sediment discharge from watersheds. N-SPECT combines data on elevation, slope, soils, precipitation, land cover characteristics, as well as surface retention and abstraction (USDA, 1986), to derive estimates of runoff, erosion and pollutant sources (nitrogen, phosphorous and suspended solids) and accumulation in stream and river networks.

Watershed boundaries for New Ireland's main island were delineated using N-SPECT based on a conditioned SRTM derived Digital Elevation Model (DEM) with 90-m spatial resolution (Appendix A). These were checked against global coastline data, and Landsat satellite imagery (2009–2013), and modified in the north-west where flat terrain prevented automatic delineation of smaller watersheds. Coastal drainage points for watersheds were determined based on this delineation and validated using existing coarse-scale stream and river data (Appendix A). Data sources and transformations for N-SPECT parameterization are described below.

2.1.2.1. Soil data. Soil data were downloaded from Version 1.1 of the Harmonized soil database of the world (Appendix A). We derived two variables for the runoff model: (i) hydrologic soil group, where soils were classified into four hydrologic soil groups (A, B, C and D) to indicate the minimum rate of infiltration obtained for bare soil after prolonged wetting (Nam et al., 2003); and (ii) soil erodibility factor (K-factor), representing soil's susceptibility to erosion by rainstorms as a function of sand, silt, clay and organic carbon concentration (see Appendix B). The average integrated K-factor was determined for each pixel using reclassification processes (Maina et al., 2012).

2.1.2.2. Rainfall data. Annual monthly average, maximum and minimum precipitation data for 2013 were obtained from Worldclim at 30 arc-seconds resolution (~1 km), and resampled to 90 m resolution. These data were used to determine the average erosive force of rainfall for each pixel, calculated from monthly rainfall data using the Modified Fournier Index (Vrieling et al., 2010) (Appendix B).

2.1.2.3. Land-use land-cover (LULC) data. A LULC classification for 2013 was derived by updating the Papua New Guinean Forestry Inventory Management System from 1996 (Appendix A Table 1) with Landsat 7 ETM+ images using on-screen digitization to distinguish forested, urbanized, and cultivated land at 100 m resolution 1 (Hansen et al., 2009), combined with further on-screen classification of oil palm plantations using maps obtained from New Britain Palm Oil. A total of 10 LULC classes were delineated. We classified established palm oil estates, new plantings (within 5 years, not yet mature), as well as independent

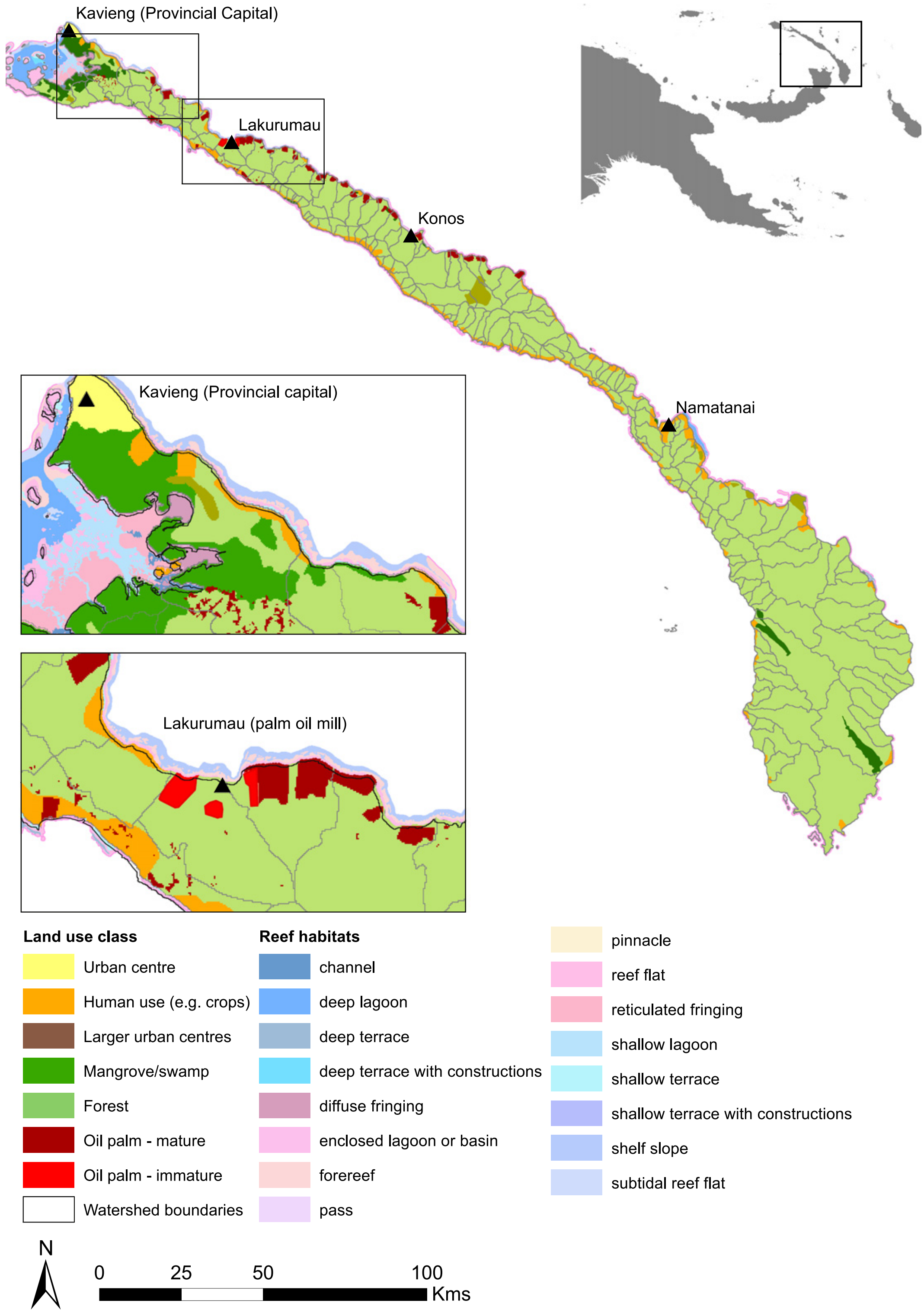

Fig. 1. Land-use map of existing land-use practices based on manual classification and onscreen digitization of Landsat 7 imagery, and (Inset) location of New Ireland Province in Papua New Guinea.

smallholders whose plantings were large enough to be delineated using satellite imagery, but exclude those smallholders whose oil palm plantings are mixed with other crops or trees that lack obvious spatial patterns necessary for their identification using satellite imagery. We cross checked the land-cover map with local observations from surveys conducted during 2013, whereby GPS was used to identify the locations of oil palm plantations along New Ireland major roads.

2.1.2.4. Development scenarios. To develop oil palm expansion scenarios and compare between sustainable and unsustainable development, we identified initial oil palm "suitability" criteria based on slope and soil constraints (Nelson et al., 2010) (Appendix B). Major companies in the oil palm industry have committed to sustainable oil palm development in Papua New Guinea following criteria designed by the Roundtable on Sustainable Palm Oil (RSPO) (http://www.rspo.org/en/how_to_

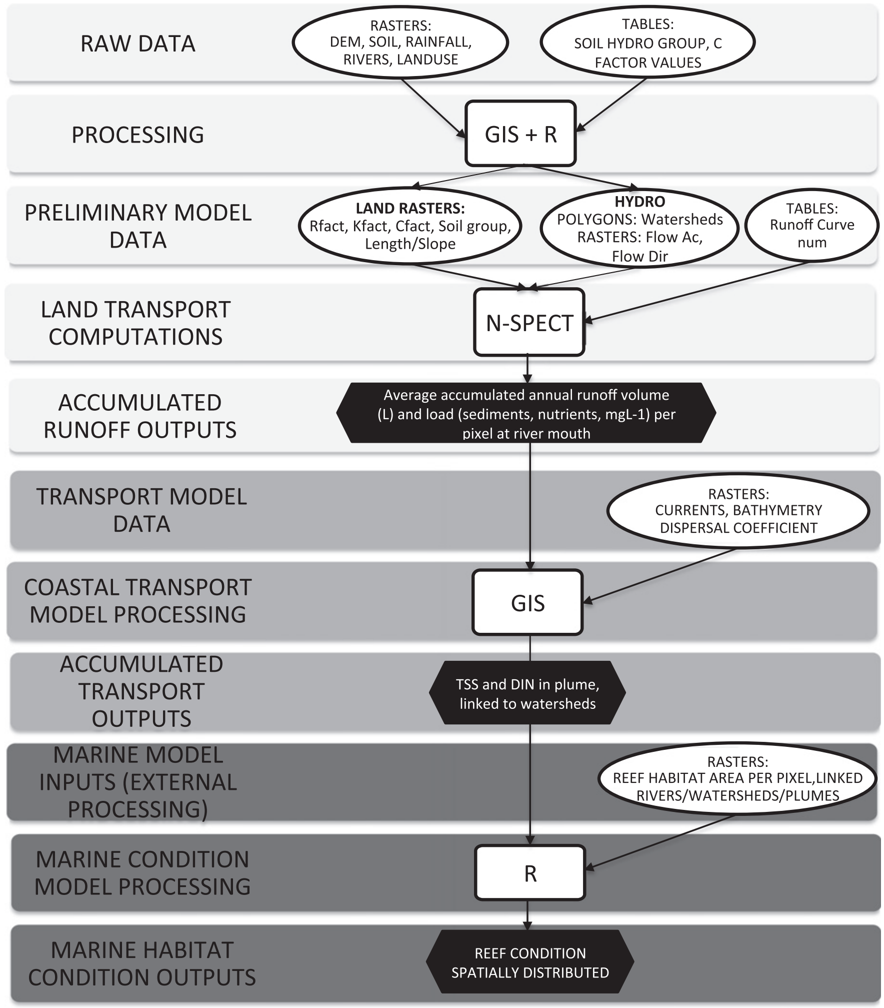

Fig. 2. Data flow diagram of the methods, and coupling processes, for the hydrological runoff (Step 1, light grey boxes), coastal sediment and nutrient dispersal and transport (Step 2, medium grey boxes), and marine ecosystem condition models (Steps 3–4, dark grey boxes).

be_rspo_certified) (Nelson et al., 2010). These consider biological productivity and environmental, social and economic factors linked with the business of palm oil production (Vis and Ng, 2005). To reduce environmental impact, RSPO prohibits clearing of high conservation value land and other fragile habitats, and recommends buffers to be created around stream and other water bodies to protect them from damage, with no new plantings on steep slopes >25°. We thus devised an RSPO development scenario that accounts for the major criteria required for RSPO accreditation. However research shows erosion increases over-proportionally (>33%) at slopes exceeding 15° (Kollert et al., 2011), compromising riparian buffer strips. We devised a new "best practice" development scenario where the maximum slope was 15°. We stress that these scenarios are not meant to predict future land-uses in any way, or to provide recommendations for sustainable development, but rather to illustrate the potential differences in soil erosion, runoff, and associated diffuse impacts on reef condition for agricultural practices that do not follow sustainability guidelines versus those that do. The following land-use scenarios were modeled (see Appendix B Fig. 2):

1. Current development – uses current land-cover and oil palm plantations (as of 2013).
2. Unsustainable development – all suitable land is converted to oil palm plantations.
3. RSPO criteria development – no primary forest conversion, riparian buffer zones (50 m); plantings on suitable soils; no conversion of peatland, mangroves or riparian regions; plantings on slopes <25°, developed close to existing roads (within 2 km); no new mills.
4. "Best practice" development – follows RSPO guidelines, but plantings only on slopes <15°, no new planting near the coast, increased riparian buffers at waterways (+100 m).

For each development scenario we converted the maximum amount of useable land to oil palm in one pulse. Although we acknowledge that in reality development would be spread across the landscape at different times, the scheduling of development was not the primary issue of concern, but scheduled development could be incorporated in future models. Large amounts of mineral fertilizer containing nitrogen and phosphorus (500 to 1000 kg/ha) are required for oil palm growth, with treatments varying over time (Comte et al., 2012). We accounted for changing levels of dissolved inorganic nitrogen (DIN) from fertilizer treatments and differing erosion potential for each stage of oil palm maturity in our model by varying pollutant coefficients for sediments and nitrogen at three stages:

Stage 1 (S1) – single pulse deforestation for oil palm in the selected areas (high fertilizer, bare land);

Stage 2 (S2) – oil palm plantations <5 years old (high fertilizer, low undergrowth);

Stage 3 (S3) – oil palm plantations at least 10 years old, established with medium density undergrowth covering regions where oil palm was planted (low fertilizer).

2.1.2.5. Modeling runoff. N-SPECT (Eslinger et al., 2005) utilizes a modified version of Revised Universal Soil Loss Equation (RUSLE) (Williams, 1975) as follows:

$$E_p = SDR_p * (R_p * K_p * SL_p * C_p) \quad (1)$$

where '*R*' is the rainfall/runoff erosivity factor per pixel '*p*', '*K*' is the soil erodibility K-factor, '*SL*' is the slope-length factor derived from the DEM, which adjusts erosion rates based on topography (Renard et al., 1997), and SDR is the sediment delivery ratio (Williams, 1977), a measure of watershed response to upland erosion which enables the model to account for retention, abstraction, and transportation of eroded soil by streams. The cover management factor (*C*) varies for each land-use type and was determined from a literature review for similar land-cover classes (see Appendix B). We changed the cover management parameter for each oil palm stage, so that new plantations have high erosion levels (bare land), versus established plantations that have less erosion (due to undergrowth and plant density) (Appendix B). To estimate sediment and nutrient concentration in river networks and at river mouths, we used flow volumes per river (L) modeled by N-SPECT to calculate concentration of total suspended sediment (TSS, mg/L) and DIN (μm/L). We also calculated sediment and nitrogen loads for pre-oil palm development land-use to determine the baseline state.

Research suggests N-SPECT is prone to over-estimating runoff loads when SDR and rainfall days parameters are not calibrated accurately (Álvarez-Romero et al., 2014). We therefore performed sensitivity analyses by running the models with varying numbers of rainfall days, and calibrated cover parameters so that soil loss rates matched those found in similar tropical catchments, whereby erosion yield from forest or oil palm agriculture was 0.001–0.1 and 2.1–40 t ha^{-1} $year^{-1}$, respectively, on flat terrain, and up to 400 t ha^{-1} $year^{-1}$ for sloping land with oil palm (Brodie and Turak, 2004; Keu, 2003; Pimentel and Kounang, 1998) (Appendix B).

2.1.3. Step 2: coastal transport modeling

To derive plume extents (the greatest monthly distance sediment particles would travel), we combined surface current velocities, bathymetry data, and soil particle settling rates (accounting for differences in particle size, texture and composition, (Hallermeier, 1981; Hill et al., 2000)), in a coastal transport model (Rude et al., 2015, Appendix B). Sediment and nutrient loading within the plumes was calculated using an exponential distance-decay function (Halpern et al., 2008). This model partially accounts for exposure (e.g., turbulence from waves), whereby the settling rate in shallow sheltered waters is an order of magnitude higher than exposed reefs (Rude et al., 2015). The combined output of these two models is TSS (mg/L) and DIN loading (uM/L) within the plumes.

2.1.4. Step 3: determine effect of fishing pressure on marine ecosystems

We estimated fishing pressure (*F*) on coastal ecosystems in New Ireland as follows:

$$F_p = \delta_h - (1 - \delta_h) e^{-\gamma_h f_p} \quad (2)$$

where f_p is the fishing intensity at pixel '*p*', derived from a combination of global artisanal fishing data (Halpern et al., 2008) and survey data on catch rates collected in the Tikana and Kavieng region (NFA, 2005), 'γ' and 'δ' are constants describing the rate of degradation for ecosystem '*h*' from fishing pressure, and minimum condition for an overfished ecosystem not subjected to watershed-based pollution respectively (Klein et al., 2012). Fine-scale survey data were not available for every pixel (NFA, 2005). Because fishing pressure on Melaniesia reefs is likely to be correlated with coastal population (Teh et al., 2009), we built a linear regression relating fishing intensity '*f*' to coastal population to extrapolate intensity across the whole province.

It is difficult to predict the impact of fishing on reef condition, with uncertainty in thresholds (Dulvy et al., 2004), trophic responses (Mumby and Harborne, 2010), and the definition of reef condition itself. We estimated 'γ' and 'δ' from the literature, improving on previous research by using relative weightings depending on ecosystem vulnerability to fish removal. For example, we applied a greater weighting to fishing impacts on coral reefs compared to seagrass (as per Halpern et al., 2008). We did sensitivity analyses varying 'γ' and 'δ' to determine degradation thresholds for each ecosystem.

2.1.5. Step 4: determine marine ecosystem condition

To predict the condition of marine ecosystems, we developed models for 9 broad coastal ecosystem types derived from the best available mapped data for New Ireland and literature values for sediment and nutrient tolerance (Appendix C). Although seagrass distribution has been mapped only at a coarse-scale, coral reef distribution data from the Millennium Coral Reef Mapping Project (Andréfouët et al., 2006) provides reef geomorphology at an increasingly fine resolution. We therefore used expert knowledge and the literature to gauge which fine-scale geomorphic classes would be likely to support seagrass habitat (Appendix C). No in-situ empirical studies have been conducted in New Ireland or neighboring regions that quantify the impact of changing runoff on coastal ecosystem condition.

We evaluated the literature to paramaterize the condition model so that it represented the likelihood of staying in a 'healthy' or high state of coral or seagrass cover, thus avoiding crossing a tipping point beyond which recovery is unlikely. The condition model accounted for direct (smothering) and indirect (light reduction) impacts from sediments, and interactions with DIN (Appendix C). We assume that at loads equivalent to pre-oil palm levels the system is in a relatively stable state (100% condition) but stressor levels higher than an estimated threshold will drive the system to a more degraded state (0% condition) (Rogers, 1990; Scheffer et al., 2001) (Table 1). We varied the parameters in the condition model by ecosystem type to account for differences in predicted species composition, depth, duration and amount of exposure to sediments and nutrients, and adaptive capacity (Browne, 2012; Dubinsky and Stambler, 1996).

Current condition (Q_t) for each ecosystem *h* at pixel '*p*' was calculated using a multiplicative risk model (Folt et al., 1999), as the product of logistic models for nutrient and sediment tolerance and fishing pressure:

$$Q_T^{hp} = \frac{e^{\left(\propto_h^S - \beta_h^S S_p\right)}}{1 + e^{\propto_h^S - \beta_h^S S_p}} * \frac{e^{\left(\propto_h^S - \beta_h^S N_p\right)}}{1 + e^{\propto_h^S - \beta_h^S N_p}} * F_p \quad (3)$$

where 'S_p' is the sediment load and 'N_p' the nitrogen load at pixel '*p*' ($p = 1…$n) given existing land-use, minus pre-oil palm levels (derived from a baseline runoff model), and 'α_h' and 'β_h' indicate the relative tolerance and rate of degradation of each ecosystem '*h*' to watershed-based pollution respectively. Condition was scaled by fishing pressure because reefs inhabited by healthy populations of fish are likely to have lower vulnerability to sedimentation (Fabricius, 2011).

The constants α and β were derived from the literature by taking estimates of sediment and nutrient tolerance thresholds (concentrations) for each ecosystem type and varying the constants until so that condition decline to <50% at the literature value for the tolerance (Table 1). Given the proximity of the fringing reefs to the coastline, we assume inshore ecosystems are pre-adapted to be less sensitive to changes in runoff loads, and so we set relatively high threshold levels for the shallow inshore ecosystems (Table 1, see also Appendix C). For wave-exposed reefs we applied an additional exposure factor whereby any sediment or nutrient loads at these ecosystems was reduced by an order of magnitude to account for expected flushing (Wolanski et al., 2005).

Condition was predicted for each development scenario by applying Eq. (3), but updating the estimates of nutrient and sedimentation loads

Table 1
Estimated thresholds of ecosystems to sediment and nutrients. We show the ecosystem likely to be supported by each geomorphic structure (seagrass or coral), the threshold bounds used for sensitivity analyses, and values used (see Table 3 Appendix C for references).

Ecosystem 'h'	Communities likely to be supported	Tolerance threshold bounds for TSS mg/L (min–max)	Tolerance threshold for DIN (μm/L)
Deep lagoon	Large deep water seagrass spp. possible, generally too deep for coral.	75–150	>7
Diffuse fringing (shallow)	Seagrass or algae, no coral. Likely *Halimeda*, possibly *Thalassia*.	75–150	>5
Shallow enclosed lagoon or basin	Seagrass or algae, primarily *Halimeda*. Likely to be soft-bottom, unlikely to have coral.	75–150	>3.5
Forereef	Coral. *Acropora* spp. and other hard corals.	50–100	1
Pinnacle	Coral only. Offshore, deeper areas.	10–50	1
Reef flat	Coral and seagrass in nearshore reef environments.	50–100	>3.5
Reticulated fringing	Coral and seagrass.	10–50	1
Shallow lagoon	Seagrass, macroalgae dominated. Likely to be soft-bottom, unlikely to have coral.	75–150	>5
Subtidal reef flat	Coral e.g. Porites spp.	30–100	1

at each oil palm development stage. Predicted condition was additionally scaled to be relative to initial condition.

2.1.6. Step 5: plan for marine conservation

Our overall objective was to maximize the chance that coastal marine ecosystems persist in New Ireland given existing oil palm plantations and different oil palm expansion scenarios. Specifically, we aimed to minimize threats to reef ecosystems by maximizing the chance that good condition areas (low risk of destruction by land runoff and fishing) were protected, subject to meeting 30% representation targets for each ecosystem in the reserve network, and keeping opportunity costs to local fishers low (Klein et al., 2013; Watts et al., 2009). Following previous studies (Klein et al., 2013), we used the inverse of our condition values as the input probability of the ecosystem being destroyed by threatening processes. To identify priorities, we used a modified version of the conservation planning software, Marxan (Version 2.4), that solves problems where there is uncertainty about the presence of a conservation feature (e.g. coral reef) due to a threat (e.g. poor water quality from run-off) (Ball et al., 2009).

We performed spatial prioritizations for each stage of oil palm growth and each scenario. Opportunity costs were determined from fishing pressure calculated in Step 3. For each scenario, we generated 100 solutions, each with a different spatial configuration, using sensitivity analyses to determine a realistic level of clumping for the priority areas (Boundary Length Modifier = 0.09). We used the selection frequency (number of times sites were selected over 100 runs) to determine the highest priority sites (those with selection frequency > 75 for all scenarios) that avoid the threat of degradation from oil palm. Finally, we evaluate the expected condition of marine ecosystems in high priority sites by re-running our models and removing the fishing parameters in areas with the highest conservation priority (>75% selection frequency).

3. Results

3.1. Land-sea runoff and response modeling

The estimates for current runoff loads were low, with land-use mapping identifying only 5622 ha of oil palm plantations across New Ireland in 2013 (Fig. 1). Under the present oil palm regime, the ecosystem models predicted <3% of coastal ecosystems were degraded (<25% of initial natural state), located mostly around the oil palm refinery on the east coast.

Differences in the amount and location of poor condition ecosystems between future scenarios were determined primarily by the intensity of oil palm development in adjacent watersheds and the stage, with newly established plantations in stage 1 causing the worst damage to reefs overall. If development was unsustainable and all available suitable land was converted to oil palm, almost 13.5 million tonnes of sediment and up to 9 thousand tonnes of nitrogen was estimated to be exported to the New Ireland coast via river discharge annually. At the watershed level, this ranged from 7 to 11,373 t year^{-1} per watershed in stage 1, averaging at 1158.7 t year^{-1}, >19 times current sediment yield (Table 2). This produced the highest overall suspended sediment concentrations in plumes ($x > 55.3$ mg L^{-1}), with over 13% of plumes containing an average TSS of >100 mg L^{-1} in stage 1, predominantly in the north and along the east coast (Fig. 3, Appendix D).

Overall, 39.2% of New Ireland reef ecosystems (193.3 km^2) are likely impacted by river discharge. Of those affected areas, almost 60% were predicted to be in poor condition (<50% of their initial state) after 5 years of unsustainable oil palm development, with only 4% of these improving once trees were mature (Fig. 3). Although RSPO criteria reduced the total area of degraded ecosystems, the spatial distribution of poor condition ecosystems was similar to the unsustainable development scenario (Fig. 4). Using the RSPO development criteria reduced average annual sediment and nitrogen loads at river mouths by 30% (257.2 T) and 7.1% (191 kg), respectively, for stage 1 development. However, watersheds dominated by oil palm typically had DIN concentrations 25 times higher than undeveloped watersheds, and produced plumes with high levels of accumulated sediments (>100 t year^{-1}). By the final stage of both RSPO and unsustainable scenarios, less than one-third of the total reef area was expected to be in a good condition (>75% of initial state) (Appendix D).

The best practice scenario produced consistently lower sediment yields per watershed ($x = 331.1$ t year^{-1}, <30% that of current yield), with lower nitrogen exports in oil palm dominated watersheds (<15% of RSPO loads) regardless of the development stage. Using more stringent development criteria also resulted in the best overall ecosystem condition, with over 60% of all coastal ecosystems in a good condition

Table 2
Average additional annual sediment and nitrogen loads at river mouths, as an increase from natural levels (prior to oil palm development) and current levels, for each development scenario and stage.

	Development scenario	Added sediment from natural levels (tons, average)	Added sediment from current (tons, average)	Added nitrogen from natural levels (kg, average)	Added nitrogen from current (kg, average)
Stage 1	Unsustainable	1158.7	1100.9	600.8	449.3
	RSPO	901.9	844.1	578.0	426.5
	Best practices	331.1	273.3	225.0	73.6
Stage 2	Unsustainable	854.4	796.6	552.5	401.1
	RSPO	667.4	609.6	529.7	378.3
	Best practices	256.9	199.1	208.0	56.6
Stage 3	Unsustainable	84.3	26.5	448.1	296.7
	RSPO	77.6	19.8	432.9	281.4
	Best practices	66.0	8.2	248.4	97.0

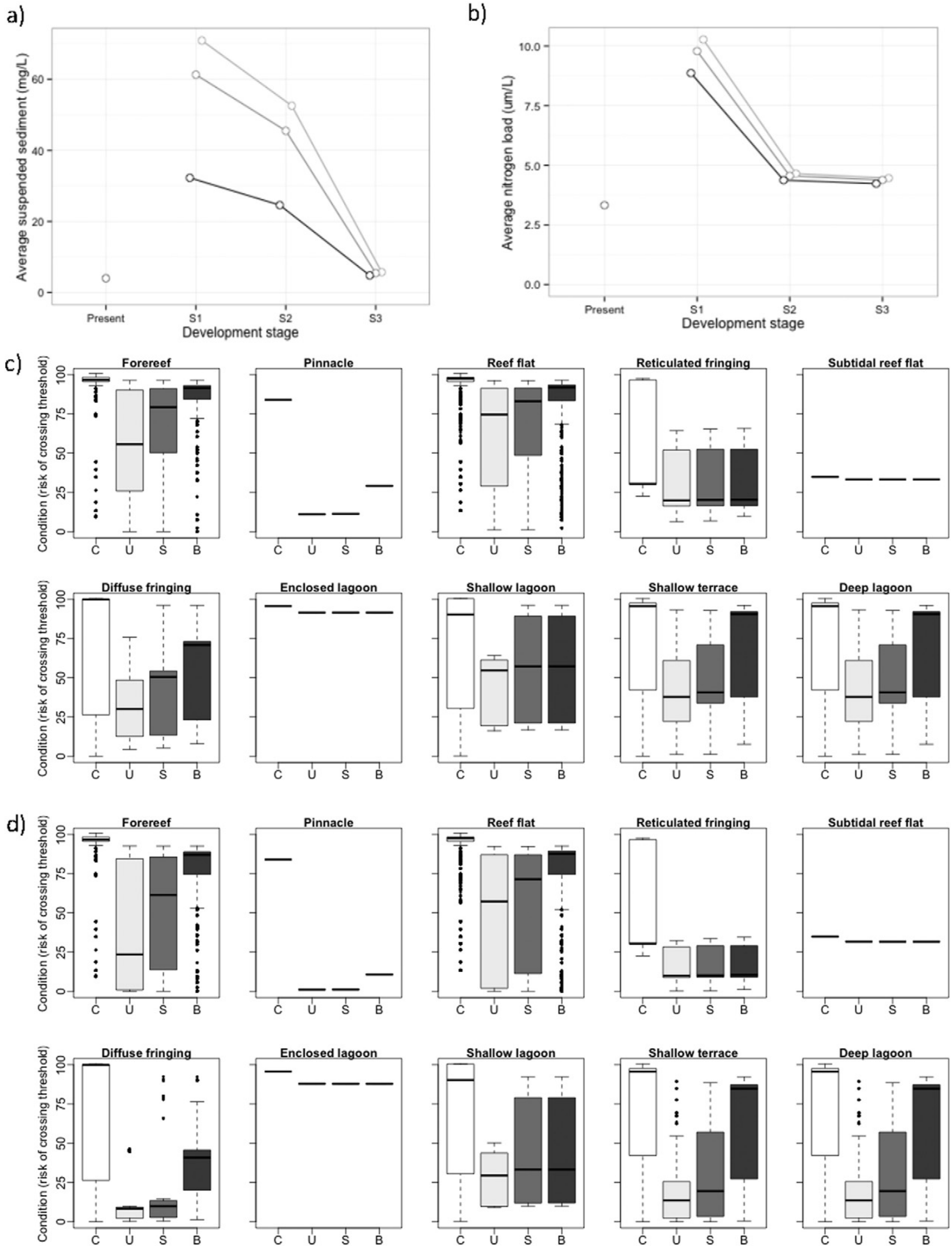

Fig. 3. Average annual suspended sediment (mg/L) in plumes (a) and average annual nitrogen load (DIN) (b) across New Ireland province, relative to natural levels, for each stage of each scenario (best practices = dark grey, RSPO = medium grey, unsustainable = light grey) for watersheds affected by increased development. Below panels identify modeled condition for ecosystems that can support coral or seagrass, for the current (C), unsustainable (U), RSPO sustainable (S) and best practices (B) scenarios, for stage 1 (c) and stage 3 (d) development, identifying median condition (black horizontal line), minimum and maximum (error bars) and 95th percentile band (shaded boxes).

regardless of the development stage (Fig. 3). Notably, best practices improved the average condition of the three seagrass ecosystems dramatically (diffuse fringing, shallow terrace and deep lagoon), with the model predicting two coral ecosystems, forereef and reef flat, would be in better condition given more stringent land development criteria (Fig. 3).

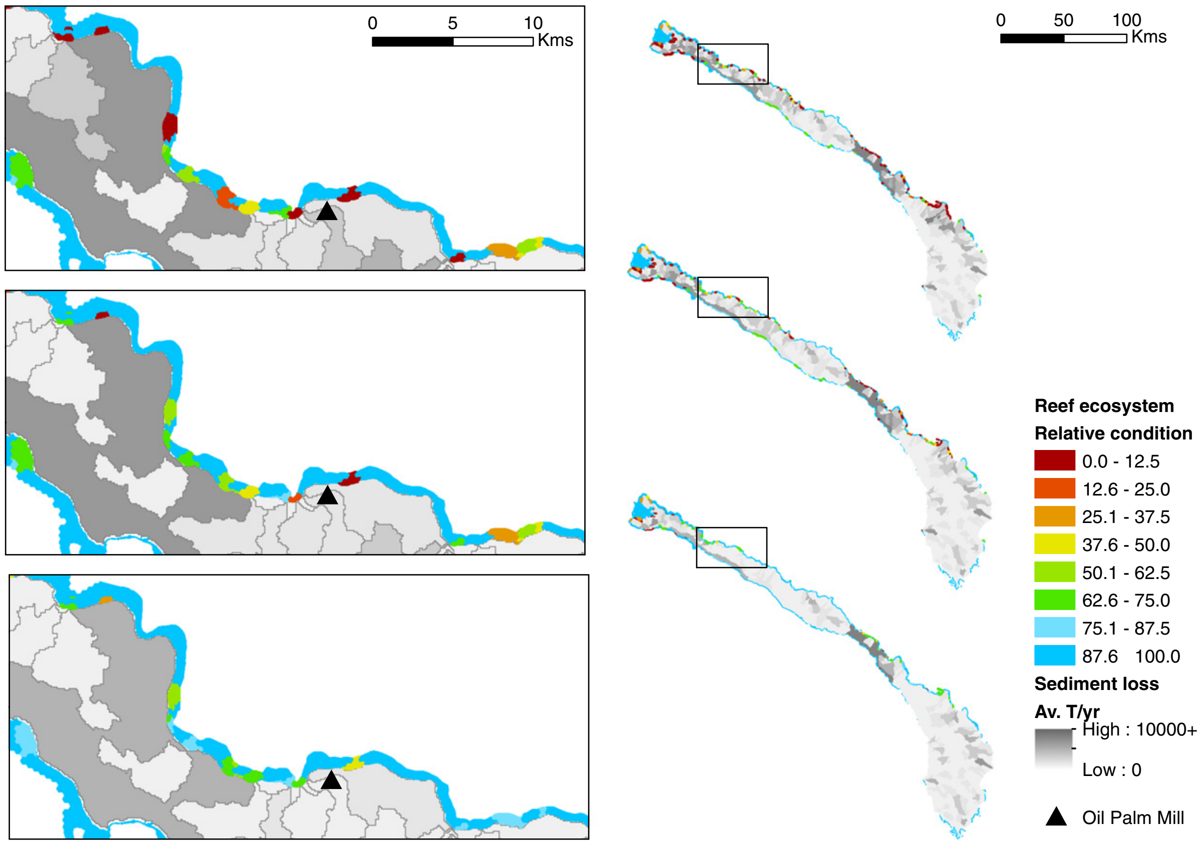

Fig. 4. Annual average sediment loss by watershed and relative condition of coastal ecosystems impacted by this runoff stage 1 development of the unsustainable oil palm land-use scenario (top maps), oil palm development following RSPO guidelines (middle maps), and development following revised best practices (bottom maps). At 100% (blue) ecosystems are in very good condition relative to the rest of the region, whilst at 0% (red) ecosystems are in very poor condition. Inset shows the full study region and area of close-up.

Over time, river loads of TSS and DIN decreased for all scenarios (Fig. 3), with a 20-fold reduction in nitrogen concentrations in runoff from stage 1 to stage 3 due to reducing the fertilizer application. Differences in erosion across watersheds were principally due to variation in watershed geomorphology, land-use, and oil palm development stage. For instance, steep slopes in the province's center produced very high river flow ($x > 300$ GL year^{-1}), with deforestation in these watersheds exporting double that of natural erosion levels to eastern reefs. Nitrogen and sediment concentrations in flood flow for these eastern rivers are up to 50 times that of rivers in the undeveloped watersheds. The heaviest river flow overall ($x = 343$ GL year^{-1}) was predicted in the mountainous south, however oil palm suitability modeling showed there is low risk of oil palm expansion there due to steep slopes and relative inaccessibility.

Several watersheds in the north exhibited consistently high sediment retention and low TSS in plumes across all scenarios despite predicted development in these regions, due to flat terrain, low erosion rates (higher clay content in soil), low delivery potential, and in some areas, filtering by mangroves, resulting in minimal change to adjacent coastal ecosystems. There was no change between scenarios, or through time, for one-third of the watersheds ($n = 170$), due to a lack of suitable soils and slopes for oil palm development in these areas. Plume extent was predicted to be restricted along much of the south and western region due to inshore currents and steep drop-offs past the reef, whilst high wave activity restricts plume extent along eastern reefs.

3.2. Priority areas for marine conservation

The marine reserve prioritization identified 7.42% of the coast consistently selected as high priority for protection (>75% selection frequency), including lagoon areas south-west of the capital Kavieng, which contain high coral biodiversity (Hamilton et al., 2009), as well as coastal areas in the center south of Konos, the north-west coast, and the southern tip of the province, due to the presence of reef ecosystems subjected to low runoff levels and fishing intensity in that region (Fig. 5a). Spatial consistencies between reserve networks were lowest when outcomes from the current scenario were compared with those from the development scenarios (Fig. 5b, also see Appendix D). Although priority areas were identified around the main town of Namatanai for current land-use, these were low to no priority once oil palm plantations expanded on the land. Similarly, areas identified as high priority around Konos given oil palm expansion were almost completely excluded if reserves were designed taking into account current land use only. Spatial consistency between reserve networks was highest for the unsustainable and RSPO scenario reserves. Several areas identified as high conservation priorities across all development scenarios were consistently predicted to be relatively high turbidity environments (average plume load >50 mg/L TSS), including areas north of Lakurumau and near Kavieng (Fig. 5).

Comparisons of reserves accounting for current and future land-use revealed up to 50% of areas prioritized for marine conservation now will contain heavily degraded ecosystem in the future if oil palm expansion continues. Some features had a high probability of destruction for the unsustainable and RSPO scenarios, in particular forereef and reef flat. Because of this, larger reserves were needed to try to meet the 30% representation target, and selection frequencies increased accordingly in these development scenarios for some moderate and high-risk areas (e.g. south-east of Kavieng, and just north of the oil palm mill in Lakurumau). Using best practices ensured there was minimal risk of future reef condition loss, as all ecosystems met their 30% representation targets and a higher proportion of good condition ecosystems were protected (Fig. 5, Appendix D). Furthermore, these reserve networks

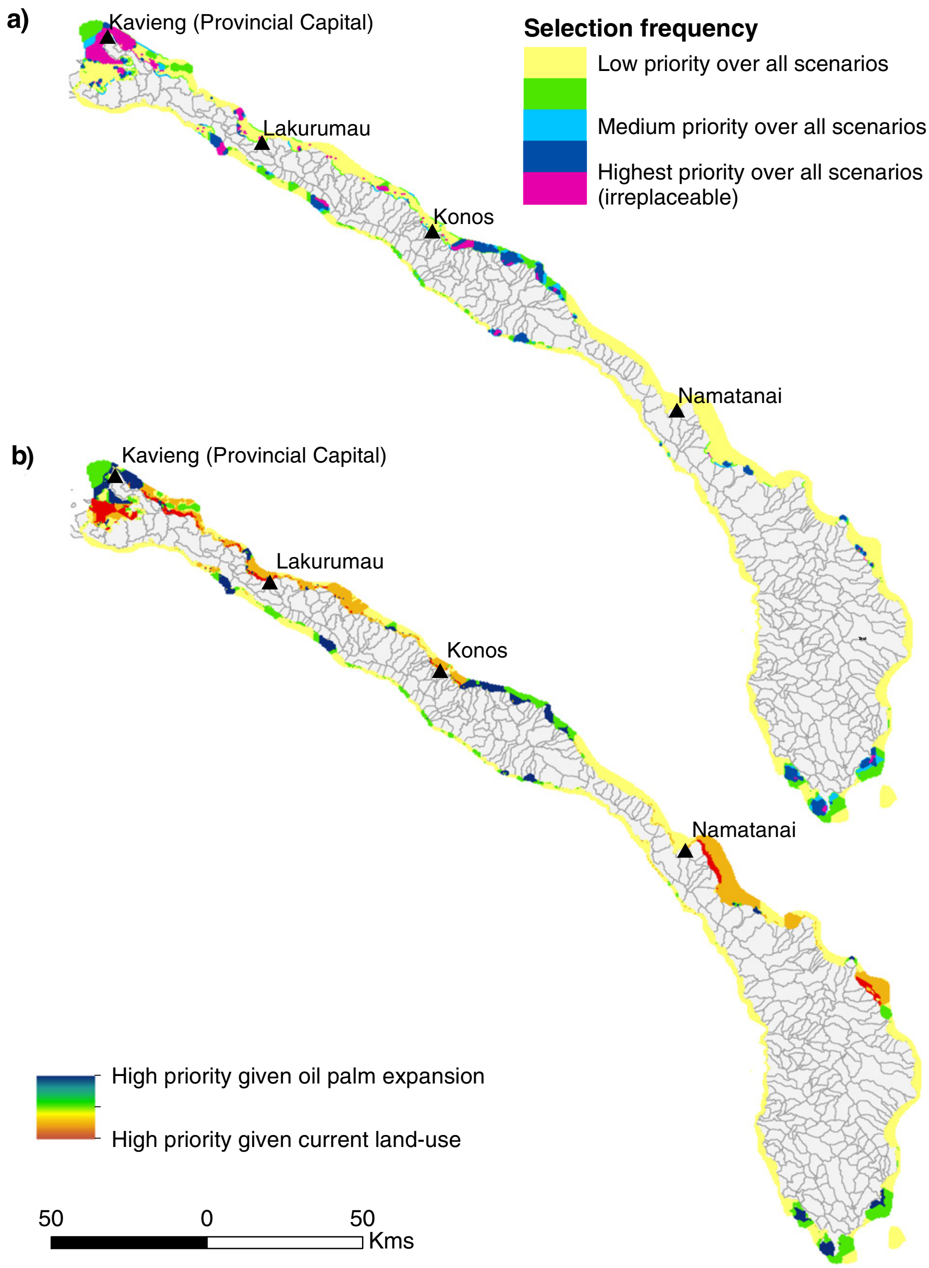

Fig. 5. (a) Map of the spatial consistency between conservation areas given the different oil palm development scenarios, taking into account future runoff impacts on reef ecosystems from oil palm development, where purple are those reserves highlighted as a priority for all scenarios, and yellow areas are rarely selected. (b) Differences between conservation areas chosen using current land-use, and those chosen given oil palm expansion, where red and orange areas are highest and medium priority respectively given current land-use, while light and dark blue areas are higher conservation priority given future oil palm development.

covered a smaller range and were up to 30% cheaper in terms of lost artisanal fishing opportunity cost than the other development scenarios.

By evaluating the expected condition of marine ecosystems in high priority conservation areas once fishing was removed, we found on average a three-fold improvement in coral reef condition for a loss of <6% of the predicted catch for local artisanal fishing. Areas where seagrass ecosystems were protected showed minimal improvement with reductions in fishing, due to the predicted minimal impact of removing fish in these ecosystems.

4. Discussion

Conservation planning for connected ecosystems at the land-sea interface is inherently difficult because planning requires linking processes for land-use change, run-off, dispersal of pollutants in the ocean and the impacts of pollutants on marine habitats (Álvarez-Romero et al., 2015a). This study linked ridge to reef processes to develop an approach for making decisions on the conservation of coral reef and seagrass ecosystems potentially threatened by expansion of oil palm plantations in a data poor region. We predict increased runoff of sediments and pollutants from unplanned oil palm expansion may substantially degrade over one-third of connected downstream ecosystems in New Ireland no matter which criteria are used to develop the land, with only minimal improvement using current guidelines for sustainable oil palm development. However, we found it is possible to substantially lower the risk of downstream ecosystem degradation by implementing new "best practices", whereby the maximum slope for plantings is reduced and development is further restricted near rivers and coasts.

Unsustainable development in the tropics has led to expansion of crops on increasingly steep slopes that contribute very high levels of sediment to rivers (Chappell et al., 2004; Sidle et al., 2006). The RSPO address land erosion in their sustainability criteria, and our model indicated RSPO guidelines reduced some of the adverse impacts from plantation development on connected marine systems. The RSPO guidelines also require protection of riparian strips which yields substantially less erosion (0.001–5 t/ha/year) overall compared to agricultural land (13–40 t/ha/year; Pimentel and Kounang, 1998) and buffers upslope runoff (Iwata et al., 2003). However our models predict that the amount of associated sediment and nitrogen exported to coastal waters using RSPO criteria may still cause significant damage to downstream ecosystems over time.

In the absence of truly sustainable oil palm development, conservation plans should account for the response of reefs to predicted levels of runoff. We identified high priority areas for conservation that consistently represent marine ecosystems that were less likely to be degraded by future oil palm development (Fig. 5a). Our results highlight the risks associated with marine conservation plans that ignore future development, with reserves containing potentially heavily degraded ecosystems if only current land-use was considered (Fig. 5b). Importantly, some areas consistently identified as a conservation priority were predicted to be turbid environments. Under traditional threat-based approaches to prioritization, these areas would likely be excluded from reserves due to the high "threat" levels (high TSS and nitrogen) (Tulloch et al., 2015). By using an approach that takes into consideration the differing sensitivities and tolerances of reef ecosystems to runoff, some of these historically low value areas were identified here as priorities for conservation. Modeling to support effective conservation decisions needs to consider ecological responses to change in threats and should not aim to just minimize threat (Giakoumi et al., 2015; Tulloch et al., 2015).

Planning of land-uses should consider downstream impacts on both coastal environments and dependent human livelihoods, such as fisheries. Oil palm is one of the most rapidly increasing crops globally (Cramb and Curry, 2012), providing income for people in many developing island nations. However, coastal communities also depend on marine resources in these regions (Allison and Ellis, 2001; Bailey and Pomeroy, 1996), and reef degradation from oil palm expansion and runoff may compromise the sustainability of coastal reef fisheries and human livelihoods (Nelson et al., 2010; NFA, 2007). Based on our findings, oil palm expansion has the potential to affect large areas of reef and seagrass across New Ireland, particularly along the densely populated north-east coast (Fig. 4). Due to the close proximity to Kavieng market, these communities depend on fisheries and other marine resources for income, with a large proportion of their income historically coming from live trade of beche-de-mer and coral trout (NFA, 2005). People living in these areas will likely be more vulnerable to increased sediments and nutrients than those living further south where there are higher runoff loads, but lower reliance on marine resources (NFA, 2007). Substantial benefits to ecosystem condition were found by protecting areas that contained ecosystems less vulnerable to runoff, at a marginal cost to fishers. Similarly, because fewer priority conservation areas were needed to meet representation targets when best practices were used, the opportunity cost of establishing reserves to fishers was reduced, highlighting the advantages of sustainable land-use practices across both land and sea ecosystems.

Our approach helps solve the dilemma of managing connected cross-system resources, reducing the risk of adverse impacts of plantation development on marine-dependent livelihoods by allowing for sustainable agricultural expansion on the land, whilst ensuring connected marine ecosystems with the highest chance of survival are adequately protected. We stress that our scenarios are not meant to predict future land-uses in any way, although the new "best practices" sustainability criteria for oil palm development may be an important consideration for future oil palm expansion in the tropics. In regions where expansion and deforestation has already occurred in upland areas, alternative management practices may be required to reduce erosion and runoff, such as building terraces, restoring heavily degraded lands, especially peatlands, and rehabilitation of mangroves, to improve the condition and build resilience of degraded or vulnerable downstream ecosystems (Comte et al., 2012; Fairhurst and McLaughlin, 2009). We also acknowledge that this study does not fully explore all criteria to reduce oil palm impact. Although RSPO accreditation requires identification and consideration of High Conservation Value habitats or areas that contain endangered species, spatial data on these features are not available for the region and were thus not considered. Similarly, though we assumed some land-use conversion in the RSPO and best-practices scenario, there are always additional infrastructure development costs that come with expanding oil palm such as road construction. Importantly, this modeling approach could be used for testing these and other criteria to make the siting of oil palm more sustainable.

There are uncertainties when making any marine conservation decision, particularly given unknown threats from the land or in limited data settings, as well as caveats with the approach and models used in this study. Our oil palm suitability model followed well-established criteria (Mantel et al., 2007) in a simple but repeatable method that used the best available data, but could be improved using more fine-scale methods such as those developed by Trangmar et al. (1995) for coffee in Papua New Guinea, or by including other factors such as crop productivity, financial viability, rights and local interests (e.g. Gingold et al., 2012). For terrestrial runoff, previous research shows models can over-estimate sediments and nutrients, particularly in steep tropical catchments (Álvarez-Romero et al., 2014). We reduced the chance of over-predicting sediment and nutrient production by using a model that accounts for retention or abstraction and performed sensitivity analyses to better match documented erosion levels from other regions (Brooks et al. 2014). Other models are available that can more accurately predict sediment loss (Sednet: Wilkinson et al., 2004), but require in situ river gauge data, which is unavailable for New Ireland. A research priority for predicting sediment from oil palm plantations should therefore be developing more accurate models, perhaps for some test regions with data, that can be used across regions with limited hydrological data. Similarly, our use of a simpler but repeatable plume modeling approach (Rude et al., 2015) accounts for several important oceanographic processes (Merritt et al., 2003), but could be improved in future studies using validation against remotely-sensed imagery classification (e.g. Álvarez-Romero et al., 2013) and ocean colour data (Schroeder et al., 2012) to more accurately calibrate pollutant dispersal in plumes. Finally, models underlying maps of fisheries pressures and costs in Papua New Guinea were relatively simple due to availability of data. However coastal communities in New Ireland vary in how much they fish and this is not always linearly related to population size (NFA, 2005). Local social surveys could improve our understanding of local fishing stressors considerably, but would necessarily come at a cost to management and potentially to biodiversity if conservation action were delayed. It would be important to weigh up the cost of collecting data against its benefits for changing the decision (Tulloch et al., 2014).

Finally, the setting of thresholds for tipping points in marine ecosystems is still highly uncertain (Selkoe et al., 2015), and further work is needed to validate models that link runoff from land-use change to coastal ecosystem condition. Although we attempted to account for general characteristics of the reef ecosystems modeled in this study when setting thresholds for degradation, the addition of cross shelf surveys and long-term empirical data on coral and seagrass cover and community assemblages would improve the accuracy of the ecosystem condition predictions, enabling more accurate calibration of tolerance thresholds and tipping points (Fabricius, 2005). However, decisions on marine conservation and oil palm are often made in data-limited countries, where reef ecosystems are already degraded, and deforestation is already occurring. Here, we provide a simple and repeatable method for cross-system planning in data-poor regions that need immediate action to prevent further biodiversity loss.

5. Conclusion

Our proposed "best practices" guidelines – no development >15° slope - should be considered for future oil palm expansion in the tropics. We illustrate the potential differences in soil erosion, runoff, and associated diffuse impacts on reef condition for agricultural practices that do not follow sustainability guidelines versus those that do. Sustainable oil palm development will reduce the impacts of pollutants on marine ecosystems, but more stringent restrictions on development are required to reduce coral degradation. Ideally, oil palm expansion must consider marine and terrestrial resource needs and inter-system connections. We recommend guidelines for sustainable oil palm development be expanded to explicitly account for ocean impacts. Finally, for decision-makers planning marine conservation at the land-sea interface, reserves designed with only existing land-uses in mind may be inadequate, and consideration of future land-use change impacts must be considered to avoid loss of marine ecosystems.

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